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Land-use and waterway quality at Mt. Grand Station, New Zealand

A thesis
submitted in partial fulfilment
of the requirements for the Degree of
Master of Applied Science

at
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by
S.M. Provost

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Abstract of a thesis submitted in partial fulfilment of the requirements for the Degree of Master of Applied Science.

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This research project focuses on the waterways of Mt. Grand, a South Island High Country sheep station. The station is 2136 ha of mostly mountainous terrain, running fine wool merino sheep and a small herd of beef cattle. Nearby, flatter land has undergone agricultural intensification, and several higher altitude areas of the station have been converted to public conservation land through Tenure Review. Situated between these conversions, Mt. Grand Station faces intensified agronomic pressures to remain economically viable, which may affect the ecological quality of its waterways. The aim of this research was to monitor sedimentation levels and phosphate concentrations (variables closely associated with land-use intensification) in three different catchments containing contrasting land-use, and to investigate how these variables affect benthic macroinvertebrate habitat. Three streams in differing catchments were sampled at different altitudes for physicochemical parameters including phosphate concentrations, visual clarity and total suspended solids. Macroinvertebrate communities were also sampled, to investigate the ecological health of each sample site. The headwaters of each stream are in steep, high altitude areas of the station, transitioning to flatter terrain at lower altitudes as they flow out to the adjacent Hawea Flat. The steep slopes of Mt. Grand face soil erosion issues, and are a ready source of sedimentation to be mobilised to the waterways. Stock have unrestricted access to all of the streams. On three occasions during the year, stream waters were sampled for analysis of Total Phosphorus, Total

Dissolved Phosphorus, cDGT and total suspended solids concentrations, as well as other associated physicochemical parameters. Total Phosphorus concentrations in riparian soils, and deposited stream sediment at each sample site were also sampled. The benthic macroinvertebrate communities were sampled to ascertain macroinvertebrate community index scores, and Ephemeroptera Plecoptera Trichoptera taxa richness percentages.

Overall, the ecological quality of stream water quality was good, but the results have shown a reduction in quality at the bottom of one catchment. Phosphate and total suspended solids concentrations were highest in a catchment containing no significant native vegetation, and increased agricultural land-use. Thick layers of deposited sediment were observed at lower altitudes of this catchment on all research trips. At the lowest altitude sample site within the catchment, Total Phosphorus concentrations in multiple samples exceeded the trigger value set for upland New Zealand streams and rivers. The Total Dissolved Phosphorus results from this sample site also indicate that a management response may be required to ensure Mt. Grand Station meets its statutory responsibilities in regards to water quality under the relevant regional plan. The sample site at the bottom of this catchment also recorded the lowest macroinvertebrate community index scores, and the lowest percentages of observed pollution intolerant taxa. The combined results from all three catchments show a negative relationship between the observed percentages of sensitive macroinvertebrate taxa, and phosphate enrichment. The findings of this research will assist station management to make better-informed decisions as to where any management responses may be best implemented, helping them to meet their statutory responsibilities in regards to maintaining the ecological quality of the regions freshwater resources. This research project paid particular focus to Mt. Grand Station because it is managed by Lincoln University.

Keywords: Freshwater management, Freshwater biodiversity, Benthic macroinvertebrates, Soil management, Phosphate concentrations, Sustainable farming.

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Chapter 1

Introduction

1.1 Rationale of Research

The loss of native biodiversity is an increasing issue of global concern as the planet's natural resources become an externality of economic and human population growth (Butchart et al. 2010; Ehrlich and Ehrlich 1981). Although biodiversity loss has occurred globally throughout all terrestrial ecosystems, the intensification of agricultural land-use is associated with many of its drivers (Hails 2002; Tscharntke et al. 2012). It is now widely recognised that unless the footprint of agricultural development is managed sustainably, both agricultural systems and natural ecosystems will incur further degradation, continuing to threaten native biodiversity and their associated ecosystem services (Chapin lii et al. 2000; Tilman et al. 2001). These concerns for native biodiversity are reiterated in New Zealand (Norton 2009; Pryde and Cocklin 1998; Saunders and Norton 2001), where biodiversity loss caused by modern agriculture is particularly worrying (Baskaran et al. 2009; MacLeod and Moller 2006; Williams and Richardson 2004). In addition, there has been an increasing realisation of the impacts that New Zealand's agricultural activities are having through diffuse pollution (Duncan 2014; Ministry for the Environment 2009). Diffuse pollution is the most extensive form of pollution, and is particularly difficult to manage and control (Novotny 2005). The source of agricultural pollution is derived from the cumulative effects of many land-uses (Lin et al. 2009), and has been identified as a major contribution to the environmental degradation of freshwater quality in New Zealand (e.g. Duncan 2014; 2017; Monaghan et al. 2007b; Smith 1993). Freshwater resources are vital for New Zealand's economic, environmental, cultural and social well-being, and also play a crucial role in supporting native biodiversity (Ministry for the Environment 2017).

In an effort to halt further ecological degradation of New Zealand's freshwater resources, the New Zealand government has implemented a National Policy Statement for Freshwater Management, which became operative in 2011. This recognises the need for policy to provide a clear national direction in freshwater

management, which it seeks to provide. The policy statement has been designed to direct the regional management of resource needs at a catchment level. This includes managing land-use and development activities, in an effort to reduce New Zealand's environmental footprint (Ministry for the Environment 2017). However, even with improvements in land-use practices and investments in mitigation, freshwater quality (in regards to ecological health and cultural values) in New Zealand is expected to continue to decline before any improvement can be made (Duncan 2014). This decline is caused by nutrient losses from previous land-use. Nutrient losses experience a lag phase whilst moving through land and water systems, before contributing to the continued eutrophication of waterways (Howden et al. 2013; Parliamentary Commissioner for the Environment 2012; 2013).

Nutrient losses from agricultural land-use are a global source of freshwater pollution, however ecosystems are often influenced concurrently by multiple stress inputs (Couillard et al. 2008; Munns and Wayne 2006; Niyogi et al. 2007). The ecological quality of freshwater resources throughout the world has been reduced through increased nutrient and fine sediment loads (Zweig and Rabeni 2001), and there is strong evidence that an increase in fine sediment is generally more detrimental to stream ecological health than an increase in nutrients (Matthaei et al. 2010; Niyogi et al. 2007; Townsend et al. 2008; Wagenhoff et al. 2011). Sediment particles can become bound with phosphorus, providing the nutrient with a method of transportation through the process of soil erosion run off, leading to an increase of in-stream phosphorus concentrations (Agudelo et al. 2011).

1.2 Aims and Objectives

The aim of this research was to assess the effects of land-use intensity on the ecological health of the waterways at Mt. Grand Station. This was established through the following research objectives:

1. To investigate if sedimentation and phosphate concentrations vary in relation to the vegetation structure of a stream catchment;

2. To investigate how sediment and phosphate concentrations affect benthic macroinvertebrate biodiversity;
3. To interpret the research findings in the context of the management of New Zealand High Country waterways.

1.3 Sites and Methods

This project focused on the freshwater resources of Mt. Grand, a 2136 ha high country station near Lake Hawea in the South Island of New Zealand that is managed by Lincoln University. Nearby the station, much of the adjacent lower elevation area of Hawea Flat has undergone agricultural intensification. Meanwhile, several higher elevation areas of Mt. Grand Station have been converted for the use of public conservation, where agricultural activities have been excluded. Situated between these conversions, Mt. Grand faces intensified agronomic pressures to remain economically viable, which may affect the ecological quality of the station's waterways. The objective of the research was to monitor Phosphate and sediment enrichment (variables closely associated with land-use intensification) of Mt. Grand's waterways, and assess how these variables affect benthic macroinvertebrate habitat availability. It also seeks to establish the significance that vegetation cover has in influencing soil erosion, which can lead to the increased sedimentation and phosphate enrichment of the stations waterways.

Mt. Grand Station is situated in the Lindis ecological district of Central Otago, and water quality management falls under the jurisdiction of the Otago Regional Council. The regional council has recognised the need to preserve its freshwater resources, and has set objective target limits for the concentrations of nitrate – nitrite nitrogen, Dissolved Reactive Phosphorus, ammoniacal nitrogen, and *Escherichia coli* in receiving waters. These concentration limits are intended to assist in achieving good water quality, and minimise the impacts of intensified land-use throughout the region. A target date of 31 March 2025 is to be set when contaminant concentrations have been found to not meet the objective targets as of 31 March 2012. These concentration limits can be found in Schedule 15 (see

Appendix A for the contaminant concentration limits that are applicable to Mt. Grand Station), which will become operative in 2020 (Otago Regional Council 2015). The three streams of focus (Grandview Creek, Lagoon Creek and Cameron Creek) in the present research project are in Receiving Water Group 3. A fourth waterway of Mt. Grand Station, Hospital Creek, had dried up during the duration of this research project and was therefore not included. The primary land-use at Mt. Grand is dryland sheep and beef farming, although various areas of notable vegetation remain on the station (Department of Conservation 2006).

Notable native vegetation of Mt. Grand Station includes a podocarp forest remnant, regenerating valley shrublands and extensive kānuka (*Kunzea ericoides*), all of which are estimated to be in pre-human condition (Department of Conservation 2006). The shrublands of Mt. Grand Station are dominated by matagouri (*Discaria toumatou*) *Coprosma* species, *Muehlenbeckia* species, bush lawyer (*Rubus cissoides*), and porcupine shrub (*Melycitus alpinus*). The invasive sweetbriar rose (*Rosa rubiginosa*) is also widespread throughout the station.

1.4 Expected Outcomes

The expected outcomes of this research are to assist farm management in making better-informed decisions regarding the freshwater resources of Mt Grand station. The research will help to establish any benefits that riparian restoration may be able to offer Mt. Grand Station in regards to helping prevent excess soil erosion and phosphate enrichment entering the streams, and what areas any such management efforts would be best to target. With the assistance of this master's project, the potential impacts that agricultural activities are having on Mt. Grand Station's freshwater resources may be reduced. It is hoped that the findings of this research project will also be beneficial to any future stream management programmes in similar upland environments.

Chapter Two

Literature Review

2.1 New Zealand Waterways

New Zealand stream ecosystems differ from their Northern Hemisphere counterparts. These differences can be attributed to the generally steep and youthful topography of New Zealand, the heavy and unpredictable nature of rainfall, a small amount of native forest in upland catchments, and the sparseness of deciduous trees (Winterbourn et al. 1981). As is common throughout other landmasses in the Southern Hemisphere, New Zealand has a low timberline at 1200 – 1500 metres. Extensive river catchments expand beyond the forest into terrain that is often steep and barren, providing a potential sediment source to waterways (Griffiths 1979).

The relatively recent arrival of humans to New Zealand has seen almost two thirds of the once near complete forest cover removed over a short historical timescale. This anthropogenic deforestation, and the subsequent changes in land-use, has vastly altered the vegetation structure of many freshwater catchments and riparian margins (Fuller et al. 2015). The depletion of catchment vegetation and immediate riparian margins has been accompanied by the degradation of freshwater ecological health (Smith 1993). Monitoring programmes throughout New Zealand have indicated the poor water quality and ecosystem health of many freshwater resources, with these issues particularly prevalent within agricultural catchments (Ballantine and Davies-Colley 2014). Early studies in New Zealand recognised the importance of land-use and riparian vegetation on stream habitat and biodiversity (Allan 1959; Phillips 1929). More recent research has shown that as a stream passes through differing intensities of land-use and the associated riparian vegetation, the structure of benthic macroinvertebrate communities is altered (Matthaei et al. 2010; Piggott et al. 2015; Wright-Stow and Wilcock 2017). Although the science of managing riparian vegetation is still relatively new, the use of riparian plantings to

create a buffer zone is now a standard best practice tool in New Zealand stream management efforts (McKergow et al. 2016).

Much of the literature on New Zealand's streams has been focused on the surrounding land-use, and the associated impacts that land-use has on stream water quality (e.g. Clapcott et al. 2016; Harding and Winterbourn 1995; Julian et al. 2017; Quinn et al. 1997a). The land-use activity that has had the greatest negative impact on freshwater quality throughout New Zealand in recent decades is high producing pasture that require large nutrient inputs to support high densities of livestock (Davies-Colley 2013; Julian et al. 2017). Freshwater invertebrates, and native and exotic fish have been shown to display a negative relationship to agricultural intensification within New Zealand stream catchments (Ramezani et al. 2016). Despite this, it has been demonstrated that stream quality can be improved when farms implement some simple mitigation practices in their management systems. Practices used to mitigate land-use impacts on New Zealand streams include preventing stock access to waterways, irrigation of dairy farm effluent and improved storage during wet conditions, riparian fencing and planting, improved grazing management, diverting run off, building bridges or culverts, improved irrigation management, the use of herd shelters or offsite grazing pads during winter, and reducing the level of Phosphorous fertiliser application (Wright-Stow and Wilcock 2017).

The relationship between land-use and chemical water quality has also been an area of research focus, which has shown significant changes associated with land-use intensification (e.g. Ballantine and Davies-Colley 2014; Hoare and Rowe 1992; Larned et al. 2016). This intensification has had a significant impact on the nutrient concentration of streams throughout New Zealand (Larned et al. 2016; Ramezani et al. 2016). Land-use intensification has also been shown to cause increased sedimentation in streams (e.g. Ballantine and Davies-Colley 2014; Cerovski-Darriau and Roering 2016; Quinn and Stroud 2002)

The risks of these negative impacts associated with land-use intensification are greatest in low-elevation catchments, where high-intensity agriculture along with

urban development are the dominant land-uses (Collier and Clements 2011; Cullen et al. 2006; Monaghan et al. 2007a). New Zealand streams of low-elevation are predominantly groundwater fed, deriving their flow from aquifers as opposed to surface runoff and interflow (Baalousha 2012; Guggenmos et al. 2011; White 2009). The groundwater sources that feed low-elevation streams are often contaminated, which then leads to an increase of in-stream contaminant concentrations. The increased concentration of contaminants is often further compounded down-stream as flows accumulate (Larned et al. 2014). Low-elevation streams and rivers of New Zealand often have problems with algal and macrophyte blooms, large diurnal fluctuations in pH, and oxygenation issues (Wilcock et al. 2007; Wilding et al. 2012).

Although the low-elevation streams of New Zealand are most at risk of water quality issues, high-elevation streams are also subject to the nutrient concentration problems that are associated with land-use intensification. High-elevation streams are the focus of the present project. A research project that sampled the sediment depositions of upland streambeds in South Otago showed greater concentrations of phosphorus in dairy farming catchments than those used for sheep farming (McDowell 2009). Results indicate that stream sediment beds may sequester excess phosphorus, which can then act as a long-term input source as sediment and phosphates are released over time, even if no further inputs occur. The sequestration of phosphates within benthic sediment can create a long-term impact on upland streams, and needs to be factored into management plans (McDowell 2009). Similar research focusing on the phosphorus concentrations of a high-elevation catchment has shown that the phosphorus export, and the effects of this export on stream waters, was generally comparable to the results of other studies focusing on lower elevation stream catchments. High-elevation catchments are often steep in nature and consist of soils that are susceptible to erosion, creating a rapid storm runoff response. In turn, this rapid runoff response during storm events creates the transportation of sediment, which is associated with elevated phosphorus concentrations. The size and frequency of storm events are thought to be the primary controlling mechanisms of phosphorus transport for streams of higher elevation. Despite storm events being primarily responsible for phosphorus

transportation; fertiliser application, livestock waste and the trampling of streambeds all contribute to the water quality issues that affect the ecological health of high-elevation streams (Caruso 2000).

2.2 Benthic Macroinvertebrate Biodiversity

One of the most widely used methods in the bio-monitoring of stream health is to assess the composition of the benthic macroinvertebrate community found utilising the freshwater habitat (Norris and Thoms 1999; Resh and McElravy 1993). The modern history of the use of indicator species as a tool for freshwater bio-monitoring began in Europe during the early twentieth century, and this idea has since become well-established (Cairns and Pratt 1993). The macroinvertebrate communities found in most streams are highly diverse, and amongst this high diversity it is hard to categorise any particular species as a keystone that is vital to the continuation of ecosystem functions (Wallace et al. 1986). However, as a group they perform essential functions and are critical to maintaining a stream's functional integrity (Angermeier and Karr 1994). Macroinvertebrates are easy to sample and identify, and different taxa display varying levels of tolerance to pollution. These qualities make macroinvertebrates particularly suitable as indicators of the health of running freshwater (Boothroyd and Stark 2000).

A number of studies have focused on benthic macroinvertebrate diversity and the impacts of different land practices (e.g. Baumgartner and Robinson 2017; Doleddec et al. 2011; Fierro et al. 2017; Genito et al. 2002; Harding and Winterbourn 1995; Larned et al. 2016; Quinn and Hickey 1990). These studies found consistent reductions in macroinvertebrate taxonomic richness within streams draining from land that is used for agriculture, in comparison to those situated in less developed catchments. However, in a comparison between catchments of tussock grassland and developed pasture, an increase in macroinvertebrate biodiversity was observed with catchment development (Riley et al. 2003). This positive response was attributed to an increase in primary production when tussock grassland was converted to pasture, leading to an increase in Phosphorous and Nitrate

concentrations. The study suggested that the positive increase observed in macroinvertebrate diversity within pastoral catchments is due to catchment development levels not exceeding the peak of nutrient subsidy effect. This effect is also known as the subsidy-stress response gradient (Odum et al. 1979), in which a potential increase in biodiversity may be observed due to elevated primary production and increased food availability for consumers at low levels of enrichment. However, once enrichment surpasses a certain threshold it may cause adverse effects. As nutrient enrichment causes the increase of algal biomass, and the fluctuation of dissolved oxygen levels, macroinvertebrate communities will become dominated by species that are tolerant of eutrophic conditions (Niyogi et al. 2007; Wagenhoff et al. 2011).

Catchment comparisons have shown the replacement of taxa sensitive to nutrient enrichment (e.g. certain Ephemeroptera, Plecoptera and Trichoptera) with taxa that are more tolerant to nutrient enrichment, where intensified land-use has occurred (Baumgartner and Robinson 2017; Genito et al. 2002; Harding and Winterbourn 1995). These changes have been attributed indirectly to modified physicochemical and biological conditions that occur in streams situated within catchments of arable land, such as elevated temperatures, flow variability, and increased nutrients leading to periphyton growth (Quinn and Hickey 1990). Ephemeroptera, Plecoptera and Trichoptera (EPT) are commonly intolerant to pollution, and their presence is widely used to indicate the disturbance of stream communities (e.g. Lenat and Crawford 1994; Ramezani et al. 2016; Wallace et al. 1996; Wright-Stow and Wilcock 2017).

In a comparison of catchments that represent four land-use types (pastoral grassland, tussock scrubland, exotic pine forest, and native beech forest), a sequential change in macroinvertebrate composition has been observed (Harding and Winterbourn 1995). This study concluded that differences in riparian vegetation type and freshwater physicochemistry are both important factors in determining the community composition of benthic macroinvertebrates. A later study focusing on the effect of land-use on riparian vegetation, ecological water quality, and macroinvertebrates drew similar conclusions (Fierro et al. 2017), finding that

anthropogenic land-use changes could be detected through a combination of physicochemical parameters of water, riparian vegetation, and macroinvertebrate assemblages. The authors suggested that macroinvertebrate communities and riparian vegetation could be particularly useful in the management and conservation of freshwater ecosystems that have been subjected to land-use changes.

The effects of shade on stream temperature can also influence the abundance of some invertebrate groups (Quinn et al. 1997a). Field observations (Quinn and Hickey 1990) and laboratory temperature tolerance studies (Quinn et al. 1994) showed that New Zealand stoneflies are sensitive to temperature increases. Increased temperatures are associated with pasture streams due to lack of vegetation cover leading to increased light availability, in comparison with forested catchments. Increased temperatures may be responsible for the lack of abundance of this group observed in streams that are situated within pastoral catchments. It is likely that several mechanisms contributed to changes in the benthic macroinvertebrate communities that were present in streams within different land-use catchments. However, those related to stream shading appear to have particular importance (Quinn et al. 1997a). Another factor that has been demonstrated to significantly impact on the benthic macroinvertebrate communities of small streams is the grazing of riparian vegetation by cattle (Conroy et al. 2016; Quinn and Hickey 1993; Quinn et al. 1992).

Benthic macroinvertebrate biodiversity is also highly susceptible to natural disturbance (Clausen and Biggs 2000; Death and Winterbourn 1995; Dewson et al. 2007; Robinson and Uehlinger 2008). Physical disturbances resulting in changes to population structure or the availability of resources, such as an increase or decrease in flow, are common in many streams (Arscott et al. 2010). These disturbances are suggested to have strong, potentially overriding influences on the community structure of benthic biodiversity (Lake and Barmuta 1986; Resh et al. 1988). A study focusing on the influence of disturbance on New Zealand streams found a strong relationship between environmental stability and the diversity of benthic macroinvertebrate communities. The patterns observed were consistent with the

idea that high diversity is maintained by an interaction between low levels of disturbance and habitat patchiness (Death and Winterbourn 1995).

A formalised methodology for the use of benthic macroinvertebrates as bio-indicators to ascertain the health of stony bottom streams in New Zealand, the Macroinvertebrate Community Index (MCI) and its quantitative equivalent (QMCI), was proposed in 1985 (Stark 1985). This methodology was then developed further to create a set of protocols for the sampling of macroinvertebrates in New Zealand's wadeable streams (Stark et al. 2001). Biotic indices such as the MCI are numerical expressions coded according to the presence of bio indicators that differ in their sensitivity to adverse environmental conditions (Graca and Coimbra 1988). The MCI assigns tolerance values to various taxa, which are based on their sensitivity to pollution and habitat disturbance. The protocols of the MCI were developed to meet the pressure from new legislation, politicians, and the general public, which sought after quality data in order to provide for good defensible interpretation and advice in regard to freshwater management.

2.3 Soil Erosion and Sedimentation

Soil erosion and stream sediment deposition has been a literature area of focus internationally (e.g. Larsen et al. 2011; Matthaei et al. 2010; Murphy et al. 2017), and locally in New Zealand (Basher 2013; Hicks et al. 1996; Phillips et al. 2017; Quinn and Stroud 2002). The scree slopes of New Zealand's mountainous regions are a continual source of sediment to be carried downstream from upper elevations (Winterbourn et al. 1981). When quantifying the sediment yields of several New Zealand rivers, Griffiths (1979) found that they were among the highest reported in the world. Erosion rates (leading to increased sedimentation) are very high in New Zealand by world standards, with around 200 megatonnes of soil being transported to the ocean each year (Hicks et al. 2011). New Zealand's steep slopes, high rates of tectonic and volcanic activity, prevalence of high-intensity rainstorms and average high rainfall all contribute to this naturally high rate of soil erosion (Hicks et al. 2011; Soons and Selby 1992). With much of New Zealand experiencing widespread

deforestation and the introduction of grazing animals in large numbers, both of these factors have contributed to accelerated rates of erosion (Glade 2003; Page et al. 2000).

New Zealand's South Island high country has been particularly subject to severe erosion issues, caused by many decades of frequent burning, deforestation, and overgrazing by sheep and rabbits (Gibbs et al. 1945). The first humans to inhabit New Zealand were Māori settlers arriving from Polynesia around 800 years ago. Māori caused widespread deforestation (around 40% of the country's forest area), especially in the east of the South Island (McGlone 1983; McWethy et al. 2009). European settlers arrived in the early 19th century and continued to clear extensive areas of vegetation (a further 30%) for timber and agriculture. Once the vegetation had been cleared, a large number of grazing animals were introduced to these altered landscapes, and within a few decades serious erosion problems became evident (Basher 2013).

Soil erosion, leading to the contamination of streams by sedimentation, is a common by-product of human activity within a catchment (Kaller and Hartman 2004). For example, agricultural land-use has been identified as a major source of sediment in New Zealand's freshwater resources (Smith et al. 1993; Vant 1999). In agricultural catchments, the increased sedimentation of streams can be attributed to point sources, such as bank erosion caused by livestock poaching, or diffusely at a larger scale caused by land-use activities such as tillage, deforestation or overgrazing (Kairis et al. 2015; Kreutzweiser et al. 2005; Larsen et al. 2009). Soil erosion and the sedimentation of streams play an important role in the phosphorous concentrations of freshwater resources. As phosphorus enters sediment, it ultimately becomes permanently deposited, or is released by various mechanisms and returned to the waterway in a dissolved form (Søndergaard et al. 2003). Phosphorus from soil enters a stream through soil erosion runoff, leading to an increase of in-stream phosphorus concentrations. Stream waterways play a major role in the transportation of sediments and nutrients (e.g. phosphorus) (Agudelo et al. 2011).

Sediment deposition is known to cause a reduction in the biomass and taxonomic richness of benthic macroinvertebrates (Quinn and Hickey 1990). In a study focusing on the short-term effects of high-suspended sediment yields on benthic macroinvertebrates, there was little evidence to link a lethal effect with exposure to high-suspended sediments over 24 hours. Although there was no direct link made with short-term high-suspended sediment exposure and a lethal effect, it was suggested that it is likely that there would be significant long-term effects on the macroinvertebrate communities of streams with high sediment loads. Rather than direct mortality, a reduction of benthic macroinvertebrate diversity is more likely to occur due to sediment deposition (Suren et al. 2005). Deposited sediment may contribute to this reduction by smothering stream substrate (Hogg and Norris 1991; Nuttall 1972; Suren and Jowett 2001) and the gills and respiratory surfaces of fish and macroinvertebrates, whilst increasing scouring, abrasion and turbidity, and changing the stratum on which periphyton can establish (Newcombe and MacDonald 1991; Ryan 1991; Waters 1995a; Wood and Armitage 1997), and reducing the quality or quantity of the periphyton food supply (Yamada and Nakamura 2002). Sediment deposition can block interstitial spaces in the benthic zone, causing a reduction in available habitat (Brunke 1999; Kaller and Hartman 2004; Ryan 1991; Waters 1995b; Wood and Armitage 1997). Increased sediment loading of a waterway also decreases photosynthesis and net productivity (Armour et al. 1991), and reduces water circulating through fine gravels, which will lead to less oxygen and an overall reduction in stream carrying capacity (Ryan 1991).

Ecosystems are often simultaneously influenced by multiple inputs (Couillard et al. 2008; Munns and Wayne 2006), and although increases in nutrients and sedimentation have both contributed to the decline freshwater ecological health, an increase in fine sediment is considered to be more detrimental than an increase in nutrients (Matthaei et al. 2010; Niyogi et al. 2007; Townsend et al. 2008; Wagenhoff et al. 2011). The results of studies on the effects of stream-stressor inputs have shown that the effects of increased sedimentation are more common than that of increased nutrients (Matthaei et al. 2010; Wagenhoff et al. 2011). It has also been demonstrated that ecological stream variables have a stronger relationship to

sediment than nutrients when investigating the multiple stressor responses in a regional set of streams (Wagenhoff et al. 2011).

2.4 Excess Nutrients

The contamination of freshwater resources from excess nutrients is a common global issue (e.g. Camargo et al. 2005; Dodds et al. 2008; Dupas et al. 2015; Huang et al. 2017; Jones 1972; Smith 2003b). As freshwater resources face increasing concentrations of excess nutrients, harmful algal blooms and cloudy waters have become a common sight throughout the world (Paerl and Otten 2013). A major driver of these problematic issues that are occurring on a global scale is the anthropogenic application of nitrogen and phosphorus (Bennett et al. 2001; Galloway and Cowling 2002; Rabalais 2002; Smith et al. 2006). Agricultural intensification has led to the increased rates of nitrogen and phosphorus being applied as fertilisers and from stock effluents, which in turn has led to the accelerated leaching of these nutrients from land to water (Smith et al. 2003; Tilman 1999). Nutrients such as nitrogen and phosphorus are susceptible to being leached from a terrestrial environment and into a waterway. Whilst Nitrate is highly soluble and readily leached from soils, phosphorus is strongly attached to soil particles and is only weakly soluble.

The formation of blooms due to the excessive growth of cyanobacteria, followed by the production of toxic compounds has been documented in many eutrophic to hypertrophic freshwater bodies throughout the world (Rastogi et al. 2014). A variety of toxic secondary compounds known as cyanotoxins have been reported from cyanobacteria blooms that are occurring in freshwater ecosystems. These toxic compounds can be highly adverse to the health and survival of aquatic organisms, wild and domestic animals, and humans (Rastogi et al. 2015). Humans and animals can be subjected to the toxic effects of cyanobacteria blooms through direct ingestion, or the consumption of water that has been contaminated by cyanotoxins (Rastogi et al. 2014). Increased nutrients can alter freshwater ecosystem functioning,

leading to toxic blooms, as growth rates in spring and summer are predominantly limited by the availability of nitrogen and phosphorous.

Nutrients such as nitrogen and phosphorus, and other agricultural chemicals, are leached from surface soil as it interacts with rainfall and runoff (Elrashidi et al. 2005). Soil nitrogen inputs become available through nitrogen fixation via legume pastures, plant residues and fertiliser application (Cameron et al. 2002; Pakrou and Dillon 2000). However, nitrogen leaching often occurs through the high concentrations of nitrogen in grazing animals' urine patches as opposed to direct fertiliser losses (Cameron et al. 2002; Di et al. 1998; Monaghan et al. 2002; Silva et al. 1999). Organic soil matter regularly contains nitrogen, but it is not available for uptake or susceptible to leaching unless it is mineralised to nitrate or ammonium (Drewry et al. 2006). During periods of low drainage volumes, pastoral systems can apply nitrogen fertiliser with no impact on waterways. However, nitrate may accumulate in the soils during these low drainage periods, to then be leached subsequently in periods of higher drainage, contributing to greater nitrogen losses (Eckard et al. 2004; Ridley et al. 2001; Tyson et al. 1997). The collective nitrate load in a waterway is of high importance, and hinges on the intensity of grazing and fertiliser use in the wider catchment (Eckard et al. 2004).

The leaching of phosphorus to waterways can occur through natural accumulation in the soil, and increased soil concentrations via the addition of fertilisers (Drewry et al. 2006). However, the loss of phosphorus to waterways will depend on the combination of source, release and transport factors (Heathwaite 2003; McDowell et al. 2004). Phosphorus has several forms, with Total Phosphorus being inclusive of Particulate Phosphorus and Dissolved Reactive Phosphorus (Drewry et al. 2006). The increased concentrations of Dissolved Reactive Phosphorus in a waterway is commonly measured by passing the water sample through a filter, which will include phosphorus that is attached to fine soil particles able to pass through the filter (Nash et al. 2002). In regards to the potential for eutrophication, Dissolved Reactive Phosphorus is considered to be fully available to aquatic plants, but only a portion of Particulate Phosphorus is available (Brodie and Mitchell 2005; Gabric and Bell 1993; McDowell et al. 2004). The use of Dissolved Reactive

Phosphorus is generally recommended for monitoring nutrient enrichment of streams and rivers (McDowell et al. 2004).

There are several methods that can be used to quantify phosphorus enrichment of freshwater resources (Benson et al. 1996; De Boer et al. 1998; Estela and Cerdà 2005; Wennrich et al. 1995; Yang et al. 2002). The conventional method to measure a water body's phosphorus content is based on a standardised fractionation scheme (APHA 2017). Another method to quantify phosphorus presence in the environment is the diffusive gradient in thin films technique (DGT) (Pichette et al. 2009), which has been applied for in-situ measurements of the phosphate enrichment of freshwater resources (Pichette et al. 2007; Zhang et al. 1998). Phosphorus species undergo dynamic interactions within water, and these interactions may alter the concentration of individual species when samples are stored for future analysis. The DGT technique allows for an in-situ measurement of Reactive Phosphorus to be taken (Pichette et al. 2009). The technique is based on a simple device that accumulates solutes on a binding agent after passage through a hydrogel, which acts as a well-defined diffusion layer (Davison and Zhang 1994). DGT is now an established method that is used in a variety of different research fields, including the assessment of water quality (Schintu et al. 2010; Sherwood et al. 2009).

With freshwater eutrophication issues being widely documented in international literature, they have also been a prominent research area of focus here in New Zealand (e.g. Ballantine and Davies-Colley 2014; McColl et al. 1977; McDowell et al. 2009; McDowell et al. 2017; Monaghan et al. 2007b; Quinn et al. 1997a; Quinn and Stroud 2002; Vant 1999). Reports describing the quality of New Zealand's freshwaters conclude that there is a strong relationship between the degree of agricultural development of catchments, and an increase in nutrient concentrations (Ballantine and Davies-Colley 2014; Quinn and Hickey 1990; Smith et al. 1993). Lowland pastoral streams bear the brunt of excess nutrient contamination (Parkyn et al. 2002). These lowland streams are typically located at the end of their catchments and owe their increased degradation to the modified hydrology of artificial drainage, decreased riparian vegetation and large quantities of contaminants being mobilised

from agricultural activities higher in the catchment (Larned et al. 2004; Wilcock et al. 1999).

The focus of the present research is the ecological quality of high-elevation stream water, but the interconnected nature of freshwater systems means that high-elevation inputs eventually reach lower altitude environments. Monitoring of lowland waterways in New Zealand has shown that many now have elevated nutrient concentrations due to agricultural land-use (Davies-Colley and Nagels 2002; Hamill and McBride 2003; Parkyn et al. 2002; Southland 2000; Wilcock et al. 1999). These impacts are particularly evident for secondary streams, where nutrient contamination levels often exceed guidelines for water quality and contact recreation (Crawford 2001). An agricultural activity of particular concern in New Zealand is dairy farming. Dairy farming has seen a pronounced change in New Zealand's agricultural landscape (Monaghan et al. 2007b). A good example of this change has occurred in the region of Southland, where dairy cow numbers increased from 25,000 in 1990, to approximately 291,000 in 2003 (LIC 1991; 2003). Streams in dairy farmed catchments in different regions throughout New Zealand have been compared with reference stream catchments of native forest composition. These comparisons have found a general degradation of stream ecological health and water quality associated with agriculture, showing that intensive dairy farming mobilises large quantities of nitrogen and phosphorus into waterways (Close and Davies-Colley 1990; Davies-Colley and Nagels 2002; Ramezani et al. 2016). Research has monitored the water quality of a New Zealand river with headwaters surrounded by sheep farming, before high intensity pasture and dairy farm occur in the mid and lower reaches (Harding et al. 1999). This showed that in comparison with the headwaters, the lower reaches of the river had a significant reduction in water clarity, an increase in benthic sediment levels, greater abundance of periphyton and an increase in nutrient concentrations. In the most detailed study comparing the effects of contrasting pastoral management systems on phosphorus losses in runoff from hill pastures, it was concluded that Total Phosphorus losses under rotational grazing management of cattle were more than twice the phosphorus losses that occurred with either rotational or continuous sheep grazing (Gillingham and

Thorrold 2000). This was predominantly due to greater sediment losses generated by cattle grazing, associated with increased treading damage and bare ground coverage.

As the effects of intensive agriculture on New Zealand's waterways became increasingly documented, the New Zealand government recognised the need to provide a clear policy for freshwater management at a national level. To achieve this, the National Policy Statement for Freshwater Management was implemented in 2011, and has since been amended multiple times to its current form that took legislative effect on 7th September 2017. The policy statement provides a National Objectives Framework to assist regional councils and communities to plan for freshwater objectives. The policy sets out national bottom lines for nutrient enrichment (and other parameters) in regards to freshwater quality, and has set a national target of 90% of specified rivers and lakes to be safe for primary contact by 2040. An interim target has been set for 80% of these rivers and lakes to be safe for primary contact by 2030 (Ministry for the Environment 2017).

2.5 Catchment Vegetation

The type of land-use and the associated vegetation structures occurring within a catchment have been shown to have a major influence on the ecological quality and community composition of streams throughout the world (e.g. Chase et al. 2016; Richards et al. 1996; Sponseller et al. 2001). The influence of catchment vegetation and land-use intensities on stream health has also been demonstrated in New Zealand (Quinn et al. 1997a; Ramezani et al. 2016; Riley et al. 2003). Converting the native vegetation structure of a stream's catchment to a different land-use can influence ecosystem health by causing changes in nutrient loading (Chase et al. 2016), solar energy fluxes (Hicks 1997), hydrology (Davies-Colley 1997), sediment inputs (Miller et al. 2011), organic matter inputs (Hicks 1997) and decomposition rates (Niyogi et al. 2003). These changes in ecosystem health are often particularly profound when a forest is converted to pasture (Hicks 1997; Verheyen et al. 2015), however they may be more subtle when converting native grassland to pasture

(Riley et al. 2003; Townsend and Riley 1999). In comparison to an undisturbed forest catchment, streams draining from pastoral land will typically display physical and chemical modification, as well as significant changes to the benthic flora and macroinvertebrate communities (e.g. Castro et al. 2017; Connolly et al. 2015; Ramezani et al. 2016; Storey and Cowley 1997; Verheyen et al. 2015).

In addition to the vegetation structure of the wider catchment, riparian vegetation plays an important role in maintaining stream ecosystem health (Connolly et al. 2015; Sweeney and Newbold 2014). The benefits of native riparian vegetation as a buffer zone have been recognised as a useful tool in mitigating the impacts of intensified land-use on streams throughout New Zealand, with significant improvements in water quality having been demonstrated in agricultural catchments through riparian management (e.g. Collins et al. 2013; Davies-Colley and Quinn 1998; Parkyn et al. 2003; Renouf and Harding 2015; Wilcock et al. 1999). As the knowledge surrounding riparian management increases, New Zealand regional councils now recommend native riparian buffer zones as a “best practice” method in efforts to improve water quality (Collins et al. 2007; Monaghan et al. 2008; Parkyn 2004; Wilcock et al. 2009).

Riparian buffer zones can have a large effect on stream water quality by filtering surface runoff, reducing in-stream sedimentation and nutrient enrichment (Fennessy and Cronk 1997; Lam et al. 2011), regulating stream temperature (Parkyn et al. 2000), stabilising banks (Marden et al. 2007), reducing peak flood flows, and providing in-stream organic matter as a food and habitat resource for fish and macroinvertebrates (Jowett et al. 2009; Parkyn 2004; Wilcock et al. 2009). The vegetation structure of the riparian zone will largely control light availability of a stream ecosystem (Davies-Colley and Quinn 1998), which is a highly influencing factor (Vannote et al. 1980). In particular, light availability will influence water temperature (Quinn et al. 1992), primary production of aquatic plants (Hill et al. 1995; Quinn et al. 1997b), solar ultraviolet exposure of stream life (Bothwell et al. 1994; Kiffney et al. 1997), and the visual behaviour of aquatic fauna (Lythgoe 1979). The management of riparian vegetation is considered to be of upmost importance

for the restoration of smaller streams in New Zealand, most of which were originally heavily shaded by native forest (Davies-Colley and Quinn 1998).

2.6 Previous Research at Mt. Grand Station

Mt. Grand was previously a Crown pastoral lease property, before undergoing a Tenure Review process that saw 530 ha of the station transferred to the Department of Conservation. Lincoln University acquired freehold ownership of the remaining 1,445 ha. Tenure Review is a process where the Crown enters into negotiations with the owners of South Island pastoral leases to redefine the associated property rights. As a result of Tenure Review, areas of pastoral estates that have high conservation and/or low agricultural value are returned to the Crown and placed under the management of the Department of Conservation. The remaining portions of the pastoral estates that are seen to have high farming values or other uses are transferred to the freehold ownership of the former lessee (Quigley 2008). As part of the Tenure Review process of Mt. Grand Station, a conservation resources report was undertaken in 2005. This conservation report found a number of threatened or uncommon flora and fauna species present at Mt. Grand. The conservation resources report also notes that several areas of remnant native vegetation on Mt. Grand are likely to be in a prehuman habitat (Department of Conservation 2006).

Previous research at Mt. Grand Station has largely been agriculturally focused (e.g. Aspinall et al. 2004; Gillespie et al. 2006; Lonati et al. 2009; Maxwell 2013; Maxwell et al. 2010; Power 2007; Power et al. 2006; Smith 2003a). Clover has been an area of particular interest for these studies. When studying the influence of environmental factors on the abundance of naturalised annual clovers in high country, soil moisture was found to be important in determining their presence or absence, and aspect was a dominant factor affecting their abundance (Maxwell et al. 2010). These results helped to further build upon the findings of earlier research, where similar rainfall for all aspects lead to the domination of sunny north facing slopes by annual clovers, and shady south facing slopes were dominated by perennial white clover (Power et al. 2006).

Research at Mt. Grand has also focused on sodium deficiency, and it has been convincingly demonstrated that the addition of coarse salt helps attract sheep to under-grazed pasture areas. After the addition of coarse salt to prior under-grazed areas of the station, a rapid increase of bare ground coverage was observed over a two-day grazing period (Aspinall et al. 2004). Building upon these findings, the application of common salt has been demonstrated to improve pasture quality and production on low producing, steep, southerly facing high country slopes (Gillespie et al. 2006). Research at the station has also provided evidence that known limitations in seedling recruitment of legumes and herbs due to insufficient grazing of competitors (Edwards et al. 2005; Lambert et al. 1985) could be overcome by the application of salt (Gillespie 2006). The effects of livestock grazing on the snow tussock grassland of Mt. Grand have also been studied (Smith 2003a). This provided evidence of a significant vegetation change occurring in the snow tussock grassland of Mt. Grand, with a decline of biomass in native species and inter-tussock species, and an increase in *Hieracium* species. The study found that appropriate stock grazing management depends on whether the objectives are production or conservation based, and will need to be site specific. Mt. Grand Station is managed by Lincoln University, hence why it is the particular focus of this research project.

Chapter Three

Methods

3.1 Overview

Three research trips were undertaken to capture seasonal variation as best as possible with the time available. Winter sampling was undertaken from 4-7 July 2016, spring sampling from 20-23 September 2016, and summer sampling from 14-17 December 2016. Stream catchments containing areas of notable remnant vegetation and lower levels of agricultural intensification were compared with a higher intensity pastoral catchment, in an effort to assess the impact that agriculture is having on the freshwater resources of Mt. Grand Station. Streams were sampled for phosphorus concentrations, dissolved oxygen, pH, conductivity and temperature, total suspended solids, and visual clarity. Visual estimations of the presence of periphyton growth were recorded, as they are indicative of excess nutrients in a waterway. The riparian vegetation structure at each site was also noted, as buffer zones of vegetation have been shown to reduce nutrient contamination of streams (Collier et al. 1995; Cooper 1990; Williamson et al. 1996).

In addition to sampling each stream for phosphate concentrations and other physicochemical parameters, benthic macroinvertebrate sampling was undertaken to build an MCI index. Nitrogen was not monitored for in the present research, as the results of preliminary monitoring showed minimal nitrogen concentrations. Sediment bound with phosphorus was considered to be the most prominent issue affecting freshwater quality at Mt. Grand Station.

3.2 Site Descriptions

Four sites were selected in the Cameron Creek and Lagoon Creek catchments, and two sites in the Grandview Creek catchment. In total, there were five Tier One sample sites, where all variables were measured. At the remaining five Tier Two sample sites DGT, total suspended solids, phosphorus in riparian soils and stream discharge were not measured (see Figure One below for a map of the research area).

Sample sites were selected for their access, their suitability to compare different land-uses, and their distance from the relevant conservation areas. At each sample site the riparian vegetation structure, stream substrate composition and periphyton growth at each sample site was described following the Stream Health Monitoring and Assessment Kit, designed to assist land-users in the monitoring of waterways (Biggs et al. 2002). The riparian vegetation structure was described for 10 x 10m stretches on each bank of each sample site, estimating the percentage cover to the nearest 5%. The stream substrate composition was described by estimating different categories of stone breadth to the nearest 10%. The abundance of periphyton growth was described in general terms, along with the colouration and length of filaments.

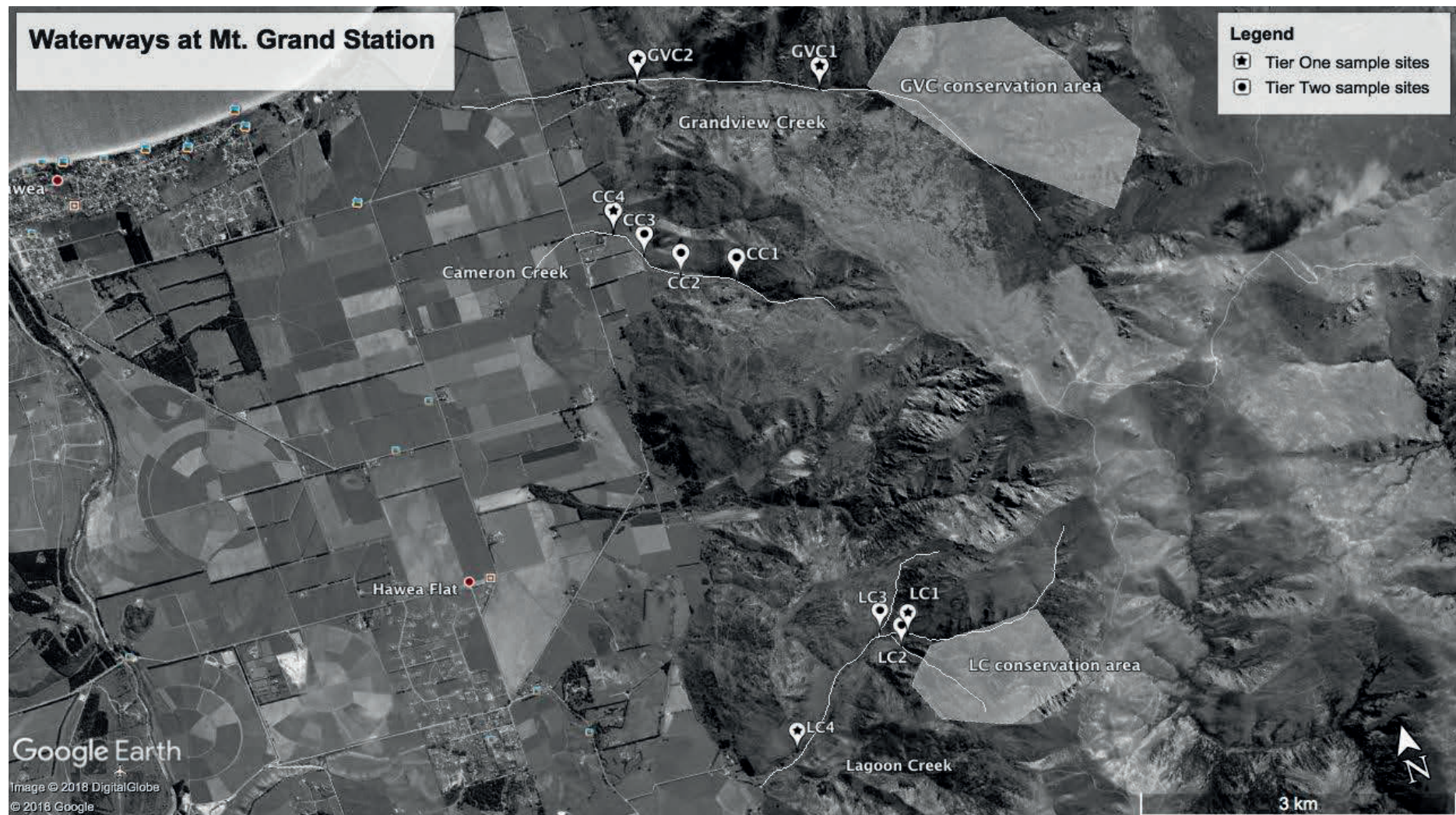


Figure One: Map of the waterways, sample sites and conservation areas that are applicable to the present research at Mt. Grand Station.

3.2.1 Grandview Creek Catchment

The majority of Grandview Creek runs on the neighbouring Lake Hawea station, however part of the catchment's upper slopes fall within the eastern boundary of Mt. Grand Station, adjacent to the Grandview Conservation area. The conservation area is located at the top of the catchment, and is 59ha in size. The top of the catchment contains a cushion field community, dense slim snow tussock and *Dracophyllum* shrublands. On the middle slopes of the catchment, significant areas of kānuka forest can be observed below the treeline. The lower slopes and valley floor are dominated by mixed shrubland and oversown and top-dressed hillside. Grandview Creek is a noticeably larger body of water in comparison to Lagoon Creek and Cameron Creek. Stocking and nutrient application rates for the Grandview Creek catchment are unknown.

3.2.2 Grandview Creek Site One (GVC1)

Tier One sample site

G.P.S. Coordinates: E2219828 N5614482

Elevation: 526m

Stream substrate: 40% boulders, 50% large cobbles and 10% small cobbles.

Periphyton growth: Small amount of green growth with short green filaments.

Riparian vegetation structure:

Left bank – 90% pasture grass and 10% introduced shrubland.

Right bank – 70% pasture grass, 10% introduced shrubland and 20% New Zealand native shrubland.

Light availability to stream: 100%

3.2.3 Grandview Creek Site Two (GVC2)

Tier One sample site

G.P.S. Coordinates: E2218312 N5614911

Elevation: 444m

Stream substrate: 20% boulders, 30% large cobbles, 30% small cobbles and 20% gravel.

Periphyton growth: Small/moderate amount of growth with long green filaments during July and September. Heavier growth observed during December research trip.

Riparian vegetation structure:

Left bank – 60% gravel, 30% pasture grass and 10% introduced shrubland.

Right bank – 60% gravel, 30% pasture grass and 10% introduced shrubland.

Light availability to stream: 100%

3.2.4 Lagoon Creek Catchment

The upper slopes of the Lagoon Creek catchment are dominated by snow tussock. Below this, a significant area of kānuka forest forms part of the middle slopes of the catchment. Encompassing both of these altitudinal areas is the Lagoon Creek conservation area. The conservation area is 118.6 ha in size, and provides an altitudinal sequence from scattered broadleaf through the kānuka forest to tussock with sub-alpine shrubland. Below this, the lower slopes of the Lagoon Creek catchment are predominantly oversown and top-dressed hillside along with mixed shrubland. Willow is present on the lower margins of the creek.

The stocking rates for the Lagoon Creek catchment were 500 ewe hoggets in the upper slopes of the catchment during the 2015-16 summer season (R. McNeilly, personal communication, December 14, 2016). There were 200 ewes at the very top of the catchment during March and May, and then the main gulley mid catchment had 500 ewes that produced 500 lambs. The December research trip was undertaken at the end of the lambing season. 125 kg/ha of Superphosphate had been applied by helicopter to the Lagoon Creek catchment prior to the research commencing.

3.2.5 Lagoon Creek Site One (LC1)

Tier One sample site

G.P.S. coordinates: E2219431 N5609443

Elevation: 558m

Stream substrate: 10% boulders, 40% large cobbles, 40% small cobbles, 10% gravel.

Periphyton Growth: Small amount of growth with short green filaments.

Riparian vegetation structure:

Left bank – 80% pasture grass, 5% introduced scrub, 5% New Zealand native shrubland, 10% New Zealand native trees.

Right bank – 90% pasture grass, 5% New Zealand native shrubland, 5% New Zealand native trees.

Light availability: 100%

3.2.6 Lagoon Creek Site Two (LC2)

Tier Two sample site

G.P.S. Coordinates: E2219433 N560944

Elevation: 563m

Stream substrate: 20% large cobbles, 70% medium cobbles, 10% gravel

Periphyton growth: None

Riparian vegetation structure:

Left bank – 30% pasture grass, 20% introduced shrubland, 30% New Zealand native shrubland, 20% New Zealand native trees.

Right bank – 40% pasture, 40% New Zealand native shrubland, 20% New Zealand native trees.

Light availability: 20%

Comments: LC2 is a smaller tributary to the main creek, running through an area of kānuka forest. The flows at LC2 were noticeably reduced in comparison with all other sites, and the flow meter could not be used here. Overhanging pasture grass and shrubs prevented the majority of sunlight from reaching the waterway.

3.2.7 Lagoon Creek Site Three (LC3)

Tier Two sample site

G.P.S. coordinates: E2219227 N5609463

Elevation: 539m

Stream substrate: 10% boulders 30% large cobbles, 50% medium cobbles, 10% gravel.

Periphyton growth: None.

Riparian vegetation structure:

Left bank – 60% pasture grass, 10% introduced scrub, 10% New Zealand native shrubland, 20% New Zealand native trees.

Right bank – 60% pasture grass, 20% New Zealand native shrubland, 20% New Zealand native trees.

Light availability: 90%

Comments: LC3 is on a larger tributary, and is located just before the confluence with the main waterway. There are no areas of notable native vegetation in this sub catchment. The area above LC3 consists of snow tussock on the upper slopes, transitioning to a mixture of oversown and top-dressed hillside and scrubland.

3.2.8 Lagoon Creek Site Four (LC4)

Tier One sample site

G.P.S. coordinates: E2218262 N5608424

Elevation: 441m

Stream substrate: 10% boulders, 30% large cobbles, 50% medium cobbles, 10% gravel.

Periphyton growth: Moderate growth with long green filaments.

Riparian vegetation structure:

Left bank – 95% pasture grass, 5% introduced shrubland.

Right bank – 90% pasture grass, 5% introduced shrubland, 5% New Zealand native shrubland.

Light availability: 100%

Comments: Willows are present upstream of LC4. The branches and debris of the willow trees seemed to be having a filtering effect on the waterway, trapping deposits of sediment.

3.2.9 Cameron Creek Catchment

The upper slopes of the Cameron Creek catchment consist of snow tussock, containing extensive areas of speargrass. The catchment's middle slopes consist of oversown and top-dressed pasture and scattered mixed shrubland, and the lower

slopes contain developed farmland extending behind the Mt. Grand Station homestead. There are no areas of notable native vegetation in the lower and middle altitude slopes of the Cameron Creek catchment.

The stocking rate for the Cameron Creek catchment over the duration of the field research was 500 ewes (R. McNeilly, personal communication, 14 December 2016). 100 tonnes of lime was applied to the catchment in 2012, and the last Superphosphate application was also in 2012. Superphosphate was to be applied to the catchment at the beginning of 2017 following the completion of field research.

3.2.10 Cameron Creek Site One (CC1)

Tier Two sample site

G.P.S. coordinates: E2218738 N5613060

Elevation: 612m

Stream substrate: 20% large cobbles, 50% medium cobbles, 30% gravel.

Periphyton growth: None.

Riparian vegetation structure:

Left bank – 5% pasture grass, 10% introduced shrubland, 85% New Zealand native shrubland.

Right bank – 5% pasture grass, 10% introduced shrubland, 85% New Zealand native shrubland.

Light availability: 10%

Comments: There is a dense pocket of New Zealand native shrubland at Cameron Creek site one, occurring near the waterway in the immediate areas above and below the sample site. The shrubs form a canopy over the sample site, which prevents the majority of sunlight from reaching the waterway.

3.2.11 Cameron Creek Site Two (CC2)

Tier Two sample site

G.P.S. coordinates: E221827 N5613200

Elevation: 511m

Stream substrate: 30% large cobbles, 50% medium cobbles, 20% gravel.

Periphyton growth: Moderate/heavy growth with long green filaments.

Riparian vegetation structure:

Left bank – 70% pasture grass, 30% New Zealand native shrubland.

Right bank – 100% pasture grass.

Light availability: 100%

Comments: Native shrubland dominated by *Discaria* and *Coprosma* species extends down from CC1 and diminishes near (upstream of) CC2.

3.2.12 Cameron Creek Site Three (CC3)

Tier Two sample site

G.P.S. coordinates: E2218005 N5613442

Elevation: 445m

Stream substrate: 20% large cobbles, 50% medium cobbles, 30% gravel.

Periphyton growth: Moderate/heavy growth with long green filaments.

Riparian vegetation structure:

Left bank – 85% pasture grass, 10% introduced shrubland, 5% New Zealand native shrubland.

Right bank – 95% pasture grass, 5% introduced shrubland.

Light availability: 100%

3.2.13 Cameron Creek Site Four (CC4)

Tier One sample site

G.P.S. coordinates: E2217796 N5613687

Elevation: 406m

Stream substrate: 10% large cobbles, 50% medium cobbles, 40% gravel.

Periphyton growth: Moderate growth with long green filaments.

Riparian vegetation structure:

Left bank – 95% pasture grass, 5% introduced trees.

Right bank – 85% pasture grass, 15% introduced trees.

Light availability: 100%

Comments: A heavy layer of deposited sediment was observed at this sample site on all research trips.

Table 1: Sample site summary.

Sample Site	Catchment	Tier One/Tier Two	Elevation
GVC1	Grandview Creek	Tier One	526m
GVC2	Grandview Creek	Tier One	444m
LC1	Lagoon Creek	Tier One	558m
LC2	Lagoon Creek	Tier Two	563m
LC3	Lagoon Creek	Tier Two	539m
LC4	Lagoon Creek	Tier One	441m
CC1	Cameron Creek	Tier Two	612m
CC2	Cameron Creek	Tier Two	511m
CC3	Cameron Creek	Tier Two	445m
CC4	Cameron Creek	Tier One	406m

3.3 Macroinvertebrates

Macroinvertebrate samples were collected from each sample site once per research trip following the protocols developed for the collection of semi quantitative macroinvertebrate data in wadeable, hardbottomed streams (Stark et al. 2001). At each sample site, riffle habitat stream substrate was disturbed using the foot-kick method (Frost et al. 1971) until a sample representative of one square metre was obtained. Immediately downstream of the foot-kick disturbance, a D-net was put in place to collect any unsettled organisms. The contents of the net were then emptied into a sorting tray and all macroinvertebrates were separated from other debris also collected in the net. Macroinvertebrates were then preserved in 70% ethanol to be identified to their genus level at a later date.

3.4 Phosphorus

3.4.1 Water Samples for Phosphorus Analysis

Samples of water were collected for Total Phosphorus and Total Dissolved Phosphorus analysis from each sample site (around 48 hours apart) twice per research trip using a 60ml syringe. Samples were taken from the thalweg at each sample site, at a depth of 0.6 x stream depth from the surface. Total Dissolved Phosphorus samples were taken with a 0.45µm filter attached to the syringe. Before the samples were collected, the syringes were rinsed twice with water from the sample site. The filters were also rinsed with 30 mL of water from the sample site. 90 ml of sample water (taken in two 45 ml amounts) was then syringed into the sample bottle. Sample bottles were then labelled and stored on ice until they could be frozen properly upon return to Lincoln University. Water samples for phosphorus analysis were defrosted and analysed at a later date using method 4500 – PJ (APHA 2017), with malachite green for the colour metric determination (Ohno and Zibilski 1991). Total Dissolved Phosphorus was analysed as opposed to Dissolved Reactive Phosphorus, as several days were spent in the field before the samples could be returned to the laboratory, and Dissolved Reactive Phosphorus samples are difficult to preserve (Jarvie et al. 2002). Sampling for Total Dissolved Phosphorus reduced the chance of sample degradation through phosphorus speciation.

3.4.2 Field Matrix Spike

Field matrix spike samples were taken to confirm that phosphorus was not lost through speciation between collecting the first samples, and analysis in the lab. For the field matrix spikes, two additional Total Phosphorus and Total Dissolved Phosphorus samples were collected as per the methods described above. One of the Total Phosphorus, and one of the Total Dissolved Phosphorus samples had 1 mL of 10 mg L⁻¹ PO₄-P solution pipetted into the sample bottle. The remaining Total Phosphorus and Total Dissolved Phosphorus samples had 1mL of 10mg L⁻¹ glucose-6-

phosphate pipetted into the sample bottle. A total of four spiked samples were taken from the first sample site on each research trip.

3.4.3 Preparation of Sample Bottles

100ml sample bottles were used to collect the water samples for Total Phosphorus, Total Dissolved Phosphorus and field matrix spikes. The sample bottles were washed out with 2% Dri-deconTM (Decon laboratories LTD, East Sussex, ENG), acid washed in 10% HCl and then rinsed with de-ionised water. After this process, the sample bottles then had an acidified persulphate reagent added, and were then ready for use in the field.

3.4.4 Diffusive Gradient in a Thin-film (DGT)

DGT probes were deployed at Grandview Creek Sites One and Two, Lagoon Creek Sites One and Four, and Cameron Creek Site Four (Tier One sample sites). The probes were deployed at the thalweg, at a depth of 0.6 x stream depth from the surface. To deploy the probes, stakes were driven into either side of the stream bank, and nylon wire was strung between them. A total of nine DGT probes were deployed at each site, composed of three sets each of three diffusion layers (0.08cm, 0.12cm and 0.16cm). The different diffusion layers were then distributed randomly across the three planar Perspex probe holders, and attached to the nylon wire strung across the stream. Once fixed in place, the DGT probes were then submerged ensuring all nine probes experienced uninterrupted and undisturbed stream flow directly across the probe sampling window to allow the diffusion of phosphate into the probe. Once the DGT probes were submerged, time and stream temperature were recorded.

Upon removing the DGT probes from a stream sample site, the time and temperature were recorded again. The DGT probes and their holders were rinsed thoroughly with de-ionised water, then carefully removed from their holders and individually rinsed again with de-ionised water, before being stored in clean, labelled plastic bags with no more than 6 DGT probes in each (DGT probes from different sites were not mixed together in storage). A small amount of de-ionised water was then sprayed into the storage bag to help prevent the DGT probes drying out. The

DGT probes were then stored inside the storage bags in a cooler bin until they could be properly refrigerated at Lincoln University before being analysed in the laboratory using the methodology developed by Jolley et al. (2016).

3.4.5 Phosphorus in Riparian Soils

The riparian soils at the same sites where DGT probes were deployed (Tier One sample sites) were sampled for the analysis of Total Phosphorus content on the first research trip. Riparian soil samples were taken 1 m from the stream edge using a soil corer. Each sample consisted of a set of three cores taken within 10cm of each other, then this triplicate was sample stored in a labelled plastic bag. This process was replicated three times on each bank of the stream sample site, with the triplicate soil cores being taken 10m from each other in a downstream direction. In total, 6 sets of soil samples (consisting of 3 cores each) were taken at each of the sample sites where DGT probes were deployed. The soil samples were then stored on ice in a cooler bin whilst in the field until they could be properly frozen at Lincoln University. The riparian soil samples were later defrosted, dried and analysed for their Total Phosphorus content using a modified EPA method 3050b (see Appendix B).

3.4.6 Phosphorus in Deposited Stream Sediment

Deposited stream sediments at all sites were sampled for the analysis of Total Phosphorus content on the second research trip. Sediment was sampled from the top 2cm of the deposited sediment layer using a scoop, collected as close to the exact sample site as possible. Sediment samples were taken downstream of the DGT probes where they were present. For some of the sample sites, sediment had to be collected further downstream (no more than 10 m from the sample site), where eddies had reduced water flow and deposited enough sediment out of suspension to allow for a sample to be collected. Sediment samples were then placed in a labelled bag and stored on ice in a cooler bin until they could be properly frozen at Lincoln University. Sediment samples were later defrosted, dried and analysed using a modified EPA method 3050b (see Appendix B).

3.5 Total Suspended Solids

Samples for the determination of total suspended solids were collected at the same sites where DGT probes were deployed (Tier One sample sites). Total suspended solids samples were collected using a 2L plastic container at the thalweg, at 0.6 x stream depth from the surface, at the time of DGT deployment. Samples were collected downstream of the DGT probes so they were not disturbed. The 2L containers were rinsed twice with stream water from the sample site, before a 2L total suspended solids sample was collected upstream of any disturbance caused. The sample containers were then labelled, before being stored on ice in a cooler bin until they could be properly frozen at Lincoln University. Samples were then defrosted and analysed using APHA method 2540D (APHA 2017).

3.6 Visual Clarity

Visual water clarity at each sample site was sampled using a 100cm visual clarity tube. The tube was used to take a 2L water sample, collected upstream from any disturbance caused when entering the water. A black magnet was then inserted into the clarity tube before sealing the lid. The black magnet was then slid down to the viewing end of the visual clarity tube, where the magnet could be viewed as it is slid back in the opposite direction. When the black magnet disappeared from view, the distance it had travelled away from the viewing end was recorded using the graduations on the side of the tube. The black magnet was then slid back in the opposite direction until it reappeared, and this distance also noted. Visual clarity readings were taken in the shade.

3.7 Stream Discharge

Stream discharge was calculated at the same sites where DGT probes were deployed (Tier One sample sites). The discharge was calculated by inserting a salt dilution into the stream, which was then monitored with an EC meter. 10L of water was mixed with 2kg of salt, and inserted upstream of the sample site at a distance of 25 x stream width. A baseline EC reference was taken on the EC meter immediately before the salt dilution was injected. The EC meter was then used to monitor the salt

dilutions flow downstream as it passed the sample site, until the baseline EC reading was again reached (Moore 2004).

3.8 Stream Flow

Stream flow rate was recorded using a stream flow meter. The flow rate was recorded at the thalweg, at 0.6 x stream depth. Maximum and average flow rates were recorded three times each, and then an average value taken.

3.9 Other Physicochemical Parameters

Temperature, pH, conductivity and dissolved oxygen were also sampled using a HACH water quality meter. These parameters were sampled twice at each site (accompanying water samples for phosphorus analysis), on each of the three research trips. For each sample, the parameters were recorded three times each at the thalweg, at 0.6 x stream depth from the surface, and then an average value taken.

3.10 Statistical Analysis

A One-way ANOVA was performed on the sample results that were normally distributed, to assess for any significant differences between each of the sample sites. A Kruskal-Wallis One Way Analysis of Variance on Ranks was performed on sample results that were not normally distributed and were unable to be normally transformed. Data was normally transformed using a Johnson's transformation. Two-way ANOVA's were performed both within and between catchments to assess for any significant differences in seasonal sampling dates for each variable. These tests and transformations were performed using SigmaPlot 12.3 (Systat Software, San Jose, CA) and Minitab® 17.2.1 (Minitab Inc., Sydney AUS). Correlation/regression analysis was undertaken to assess the relationships between variables using Microsoft Excel 2016 (Microsoft Corporation, Redmond, USA). Principal components analyses were undertaken using Minitab.

Chapter Four

Results

4.1 Macroinvertebrates

4.1.1 Macroinvertebrate Community Index Scores

Macroinvertebrate community index (MCI) scores varied significantly between sample sites (Figure 2) when analysed using a One-way ANOVA . This variation was most notable at CC4, which had significantly different MCI scores in comparison with GVC1 ($p=0.006$), GVC2 ($p=0.003$), LC1 ($p=0.002$), LC4 ($p=0.016$) and CC1 ($p<0.001$). MCI scores at CC3 varied significantly in comparison with GVC2 ($p=0.048$), LC1 ($p=0.030$), LC3 ($p=0.005$) and CC1 ($p=0.010$). LC2 varied significantly in comparison with LC3 ($p=0.021$) and CC1 ($p=0.040$), and CC2 was significantly different in comparison with CC1 ($p=0.045$). The lowest macroinvertebrate community index score was observed at CC4. Two-way ANOVA analysis did not show a significant difference in MCI scores over the different seasonal sampling dates when comparing the three catchments. There was a significant difference over the different seasonal sampling dates at sites within the Grandview Creek catchment ($p=0.032$). No significant difference was observed over the different seasonal sampling dates at sites within the Lagoon Creek and Cameron Creek catchments.

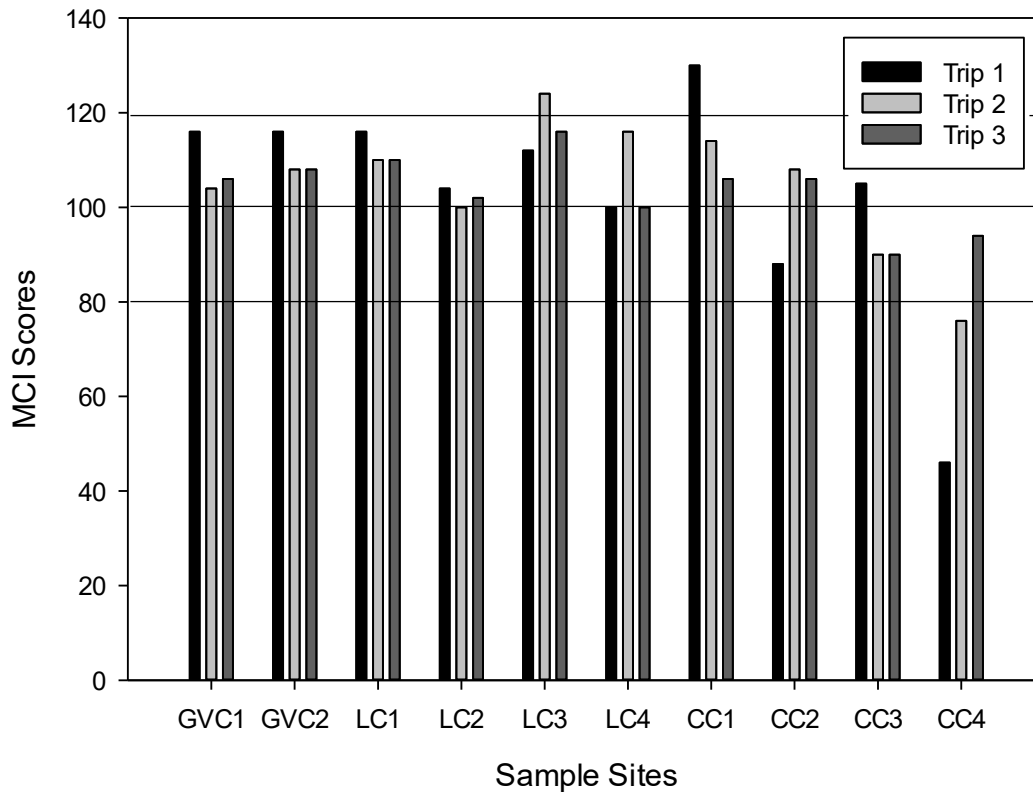


Figure 2: Macroinvertebrate community index (MCI) scores at each sample site, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. Each bar represents the MCI score for a single sampling event. The horizontal lines represent the interpretation of the MCI index, with scores of >119 classed as excellent, 100-119 classed as good, 80-99 classed as fair and <80 classed as poor. Samples were collected over the duration of three research field trips and the data combined.

4.1.2 Percentage of Macroinvertebrate EPT Taxa Richness

The percentage of observed macroinvertebrate EPT taxa at CC4 varied significantly in comparison with other sample sites (Figure 3) when analysed using a One-way ANOVA. Observed percentages at CC4 were significantly different to GVC1 ($p=0.014$), LC1 ($p=0.004$), LC2 ($p=0.001$), LC3 ($p=0.025$), LC4 ($p=0.020$), CC1 ($p=0.003$), CC2 ($p=0.028$) and CC3 ($p=0.033$). The lowest percentage of EPT taxa was observed at CC4. Two-way ANOVA analysis did not show a significant difference in percentage of macroinvertebrate EPT taxa over the different seasonal sampling dates when comparing the three catchments. There was no significant difference over the different seasonal sampling dates at sites within the Grandview Creek and Cameron Creek catchments. A significant difference was observed at sites within the Lagoon Creek catchment ($p=0.003$).

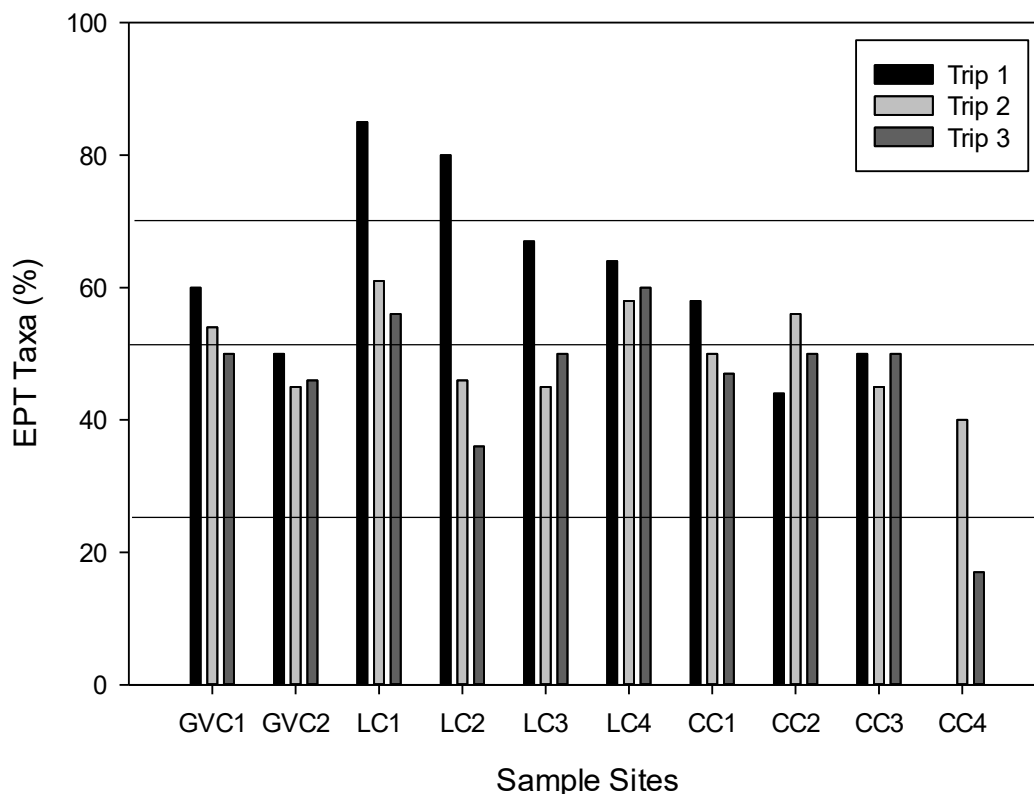


Figure 3: Percentages of observed macroinvertebrate EPT taxa richness at each sample site, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. Each bar represents the EPT taxa percentage for a single sampling event (CC4 had 0% on Trip 1). The horizontal line represents the interpretation of EPT taxa percentages, with percentages >70 classed as excellent, 51-70 classed as good, 25-50 classed as fair and <25 classed as poor. Samples were collected over the duration of three research fieldtrips and the data combined.

Table 2: The order (and number) of macroinvertebrates recorded at each sample site on trip one (see Appendix C for complete macroinvertebrate total abundance data).

Sample Site	Order (Number of Individuals Recorded)
GVC1	Coleoptera (2), Diptera (18), Ephemeroptera (20), Littorinimorpha (2), Megaloptera (13), Plecoptera (4), Trichoptera (51).
GVC2	Diptera (18), Ephemeroptera (67), Megaloptera (2), Oligochaeta (7), Trichoptera (53).
LC1	Ephemeroptera (79), Oligochaeta (2), Plecoptera (16), Trichoptera (31), Tricladida (2).
LC2	Ephemeroptera (13), Oligochaeta (6), Plecoptera (1), Trichoptera (6).

LC3	Coleoptera (3), Diptera (3), Littorinimorpha (25), Oligochaeta (6), Plecoptera (9), Tricoptera (19).
LC4	Coleoptera (1), Diptera (1), Ephemeroptera (65), Littorinimorpha (1), Oligochaeta (1), Plecoptera (16), Tricoptera (68).
CC1	Coleoptera (1), Diptera (1), Tricladida (3), Ephemeroptera (28), Oligochaeta (9), Plecoptera (6), Trichoptera (6),
CC2	Diptera (2), Tricladida (1), Littorinimorpha (34), Mecoptera (1), Oligochaeta (2), Plecoptera (6), Trichoptera (6).
CC3	Diptera (4), Ephemeroptera (4), Littorinimorpha (18), Plecoptera (16).
CC4	Diptera (2), Littorinimorpha (5), Oligochaeta (3).

Table 3: The order (and number) of macroinvertebrates recorded at each sample site on trip two (see Appendix C for complete macroinvertebrate total abundance data).

Sample Site	Order (Number of Individuals Recorded)
GVC1	Coleoptera (3), Diptera (88), Ephemeroptera (29), Littorinimorpha (17), Oligochaeta (1), Trichoptera (41).
GVC2	Coleoptera (1), Diptera (126), Ephemeroptera (129), Megaloptera (1), Oligochaeta (3), Trichoptera (65).
LC1	Diptera (25), Ephemeroptera (21), Littorinimorpha (3), Megaloptera (2), Oligochaeta (1), Plecoptera (5), Tricoptera (47), Tricladida (1).
LC2	Coleoptera (4), Diptera (8), Ephemeroptera (44), Oligochaeta (1), Plecoptera (30), Trichoptera (5),
LC3	Coleoptera (28), Diptera (9), Ephemeroptera (8), Littorinimorpha (85), Oligochaeta (2), Plecoptera (18), Trichoptera (24), Tricladida (2).
LC4	Coleoptera (4), Diptera (21), Ephemeroptera (226), Oligochaeta (7), Plecoptera (6), Trichoptera (81).
CC1	Diptera (1), Ephemeroptera (36), Oligochaeta (3), Plecoptera (1), Trichoptera (8).
CC2	Coleoptera (9), Ephemeroptera (26), Littorinimorpha (156), Plecoptera (1), Trichoptera (3), Tricladida (3).
CC3	Coleoptera (2), Diptera (32), Ephemeroptera (10), Littorinimorpha (105), Oligochaeta (2), Plecoptera (47), Trichoptera (3).
CC4	Ephemeroptera (3), Littorinimorpha (129), Oligochaeta (31), Trichoptera (1).

Table 4: The order (and number) of macroinvertebrates recorded at each sample site on trip three (see Appendix C for complete macroinvertebrate total abundance data).

Sample Site	Order (Number of Individuals Recorded)
GVC1	Diptera (30), Ephemeroptera (14), Littorinimorpha (34), Megaloptera (7), Oligochaeta (3), Plecoptera (6), Trichoptera (50), Tricladida (1).
GVC2	Coleoptera (1), Diptera (18), Ephemeroptera (5), Littorinimorpha (6), Megaloptera (1), Oligochaeta (4), Trichoptera (22).
LC1	Coleoptera (7), Diptera (12), Ephemeroptera (52), Littorinimorpha (6), Megaloptera (4), Oligochaeta (3), Plecoptera (6), Trichoptera (28).
LC2	Coleoptera (4), Diptera (3), Ephemeroptera (57), Oligochaeta (10), Plecoptera (3), Trichoptera (4).
LC3	Coleoptera (5), Diptera (15), Ephemeroptera (11), Littorinimorpha (12), Plecoptera (1), Trichoptera (11), Tricladida (3).
LC4	Coleoptera (6), Diptera (6), Ephemeroptera (60), Oligochaeta (4), Plecoptera (1), Trichoptera (67),
CC1	Archaeocopida (2), Coleoptera (3), Diptera (3), Ephemeroptera (25), Littorinimorpha (9), Oligochaeta (1), Plecoptera (3), Trichoptera (6), Tricladida (1).
CC2	Coleoptera (5), Diptera (12), Ephemeroptera (79), Littorinimorpha (104), Oligochaeta (4), Plecoptera (1), Trichoptera (15), Tricladida (9).
CC3	Coleoptera (1), Diptera (11) Ephemeroptera (38), Littorinimorpha (91), Oligochaeta (5), Plecoptera (24), Trichoptera (16), Tricladida (2).
CC4	Coleoptera (1), Diptera (1), Ephemeroptera (1), Littorinimorpha (22), Oligochaeta (2).

4.2 Phosphorus

4.2.1 Total Phosphorus

In stream Total Phosphorus concentrations varied significantly between sample sites (Figure 4) when analysed using a One-way ANOVA. This variation was most notable at CC4, which had significantly different Total Phosphorus concentrations in comparison with GVC1 ($p<0.001$), GVC2 ($p<0.001$), LC1 ($p<0.001$), LC3 ($p=0.006$), LC4 ($p<0.001$) and CC2 ($p=0.020$). Total Phosphorus concentrations at LC2 displayed significant differences in comparison with GVC1 ($p<0.001$), GVC2 ($p=0.002$), LC1 ($p<0.001$) and LC4 ($p=0.016$). LC3 was significantly different in comparison with LC1 ($p=0.005$), and LC4 was significantly different in comparison with LC1 ($p=0.013$). CC1 was significantly different in comparison with GVC1 ($p=0.001$), GVC2 ($p=0.005$), LC1 ($p<0.001$) and LC4 ($p=0.042$). CC2 was significantly different in comparison with GVC1 ($p=0.023$) and LC1 ($p=0.001$). CC3 was significantly different in comparison with GVC1 ($p=0.005$), GVC2 ($p=0.018$) and LC1 ($p<0.001$). With the exception of LC2, the highest concentrations of Total Phosphorus for each catchment were at the lower elevations. The highest concentrations of Total Phosphorus were observed at CC4. Two-way ANOVA analysis did not show a significant difference in Total Phosphorus concentrations over the different seasonal sampling dates when comparing the three catchments. There was a significant difference over the different seasonal sampling dates at sites within the Grandview Creek ($p=0.012$) and Lagoon Creek ($p=0.012$) catchments. No significant difference was observed over the different seasonal sampling dates at sites within the Cameron Creek catchment.

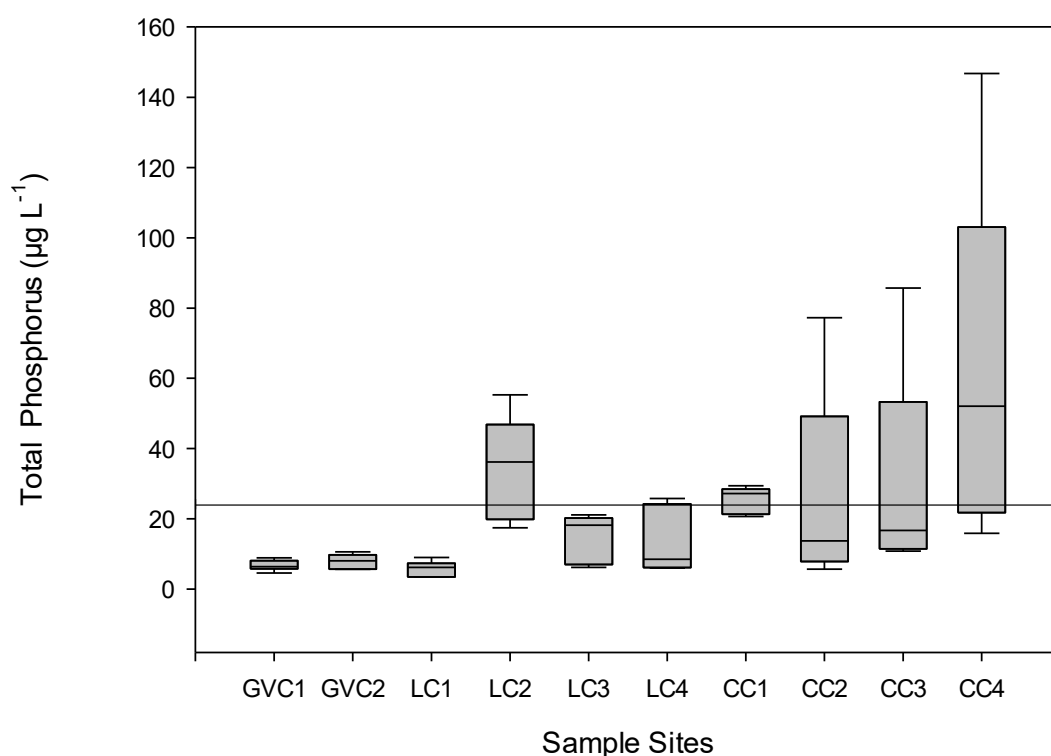


Figure 4: Total Phosphorus concentrations in stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with the median in the centre. The whiskers show the maximum and minimum values. The horizontal line represents the 24 $\mu\text{g L}^{-1}$ Total Phosphorus trigger value, which is intended to trigger a management response in New Zealand upland streams. Samples were collected over the duration of three research field trips and the data combined.

4.2.2 Total Dissolved Phosphorus

In stream Total Dissolved Phosphorus concentrations at CC4 varied significantly in comparison with other sample sites (Figure 5) when analysed using a One-way ANOVA. CC4 was significantly different to GVC1 ($p=0.008$), GVC2 ($p=0.001$), LC1 ($p=0.034$) and LC4 ($p=0.015$). The highest concentrations of Total Dissolved Phosphorus were observed at CC4. Two-way ANOVA analysis showed a significant difference in Total Dissolved Phosphorus concentrations over the different seasonal sampling dates when comparing the three catchments ($p<0.001$). There was a significant difference over the different seasonal sampling dates at sites within the Grandview Creek ($p=0.014$) and Lagoon Creek ($p=0.011$) catchments. A significant interaction between date and Total Phosphorus concentration was unable to be tested for at sites within the Cameron Creek catchment.

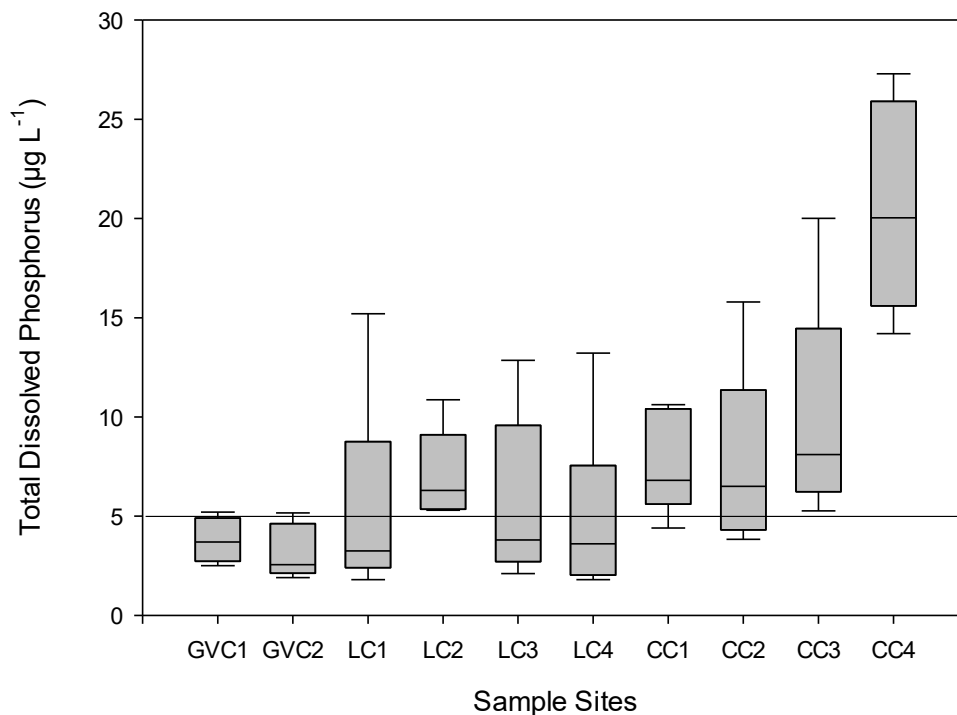


Figure 5: Total Dissolved Phosphorus concentrations in stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with the median in the centre. The whiskers show the maximum and minimum values. The horizontal line represents the 5 $\mu\text{g L}^{-1}$ Dissolved Reactive Phosphorus (a component of Total Reactive Phosphorus) trigger value, which is intended to trigger a management response in New Zealand streams. Samples were collected over the duration of three research field trips, and the data combined.

4.2.3 Diffusive Gradient in a Thin-film (DGT)

Analysis of the DGT sample results did not show any significant differences between sample sites (see Appendix D for box and whisker plot) when analysed using a One-way ANOVA. DGT samples at LC4 provided no data for two of the research trips and appear not to have worked, but there was no apparent reason for this. Two DGT samples from GVC2 were below the method detection limit ($1.5 \mu\text{g L}^{-1}$). The highest concentrations of cDGT were observed at CC4. Two-way ANOVA analysis did not show a significant difference in cDGT concentrations over the different seasonal sampling dates when comparing the three catchments. There was no significant difference in cDGT concentrations over the different seasonal sampling dates when comparing sites within the Grandview Creek catchment. Significant differences over the different seasonal sampling dates in the Lagoon Creek and Cameron Creek

catchments could not be tested for due to missing data not allowing the use of a Two-way ANOVA.

4.2.4 Phosphorus in Riparian Soils

The concentrations of Total Phosphorus in the riparian soils varied significantly between sample sites (Figure 6) when analysed using a One-way ANOVA. This variation was most notable at CC4, where the riparian soil contained significantly different phosphorus concentrations to GVC1 ($p < 0.001$), GVC2 ($p < 0.001$), LC1 ($p < 0.001$) and LC4 ($p < 0.001$). GVC1 was also significantly different to GVC2 ($p = 0.045$) and LC4 ($p = 0.006$). Significant differences were also observed between LC1 and GVC2 ($p = 0.047$), and LC4 ($p = 0.006$). The highest concentration of Total Phosphorus in the riparian soils was observed at CC4. Significant differences over the different sampling dates were not tested for, as riparian soil samples were only taken on one sampling date.

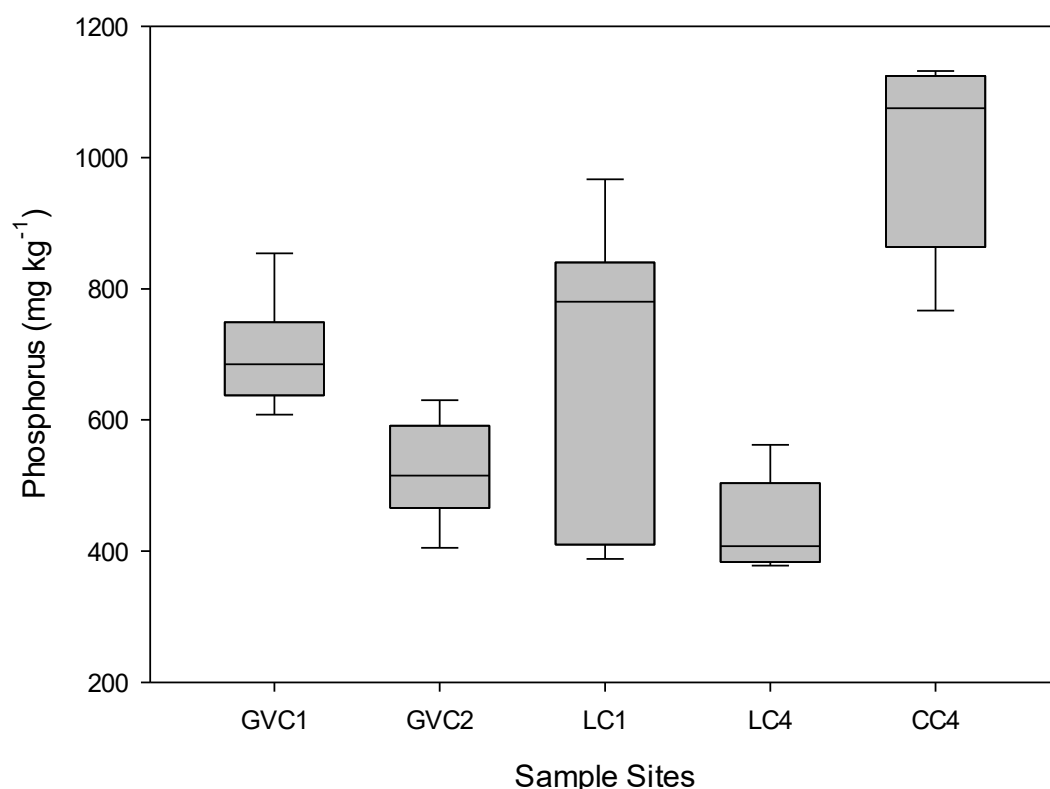


Figure 6: Total Phosphorus concentrations in the riparian soil at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with the median in the centre. The whiskers show the maximum and minimum values. Three samples were taken at 10m intervals on the left and right bank of each sample site within 1m of the stream edge during the first research trip, and the data combined.

4.2.5 Phosphorus in Deposited Stream Sediment

The Total Phosphorus concentrations in deposited stream sediment were highest at sample sites situated at the bottom of each catchment (Figure 7) when analysed using a One-way ANOVA. The highest concentration of Total Phosphorus in stream sediment was observed at LC4. Significant differences over the different sampling dates were not tested for, as stream sediment samples were only taken on one sampling date.

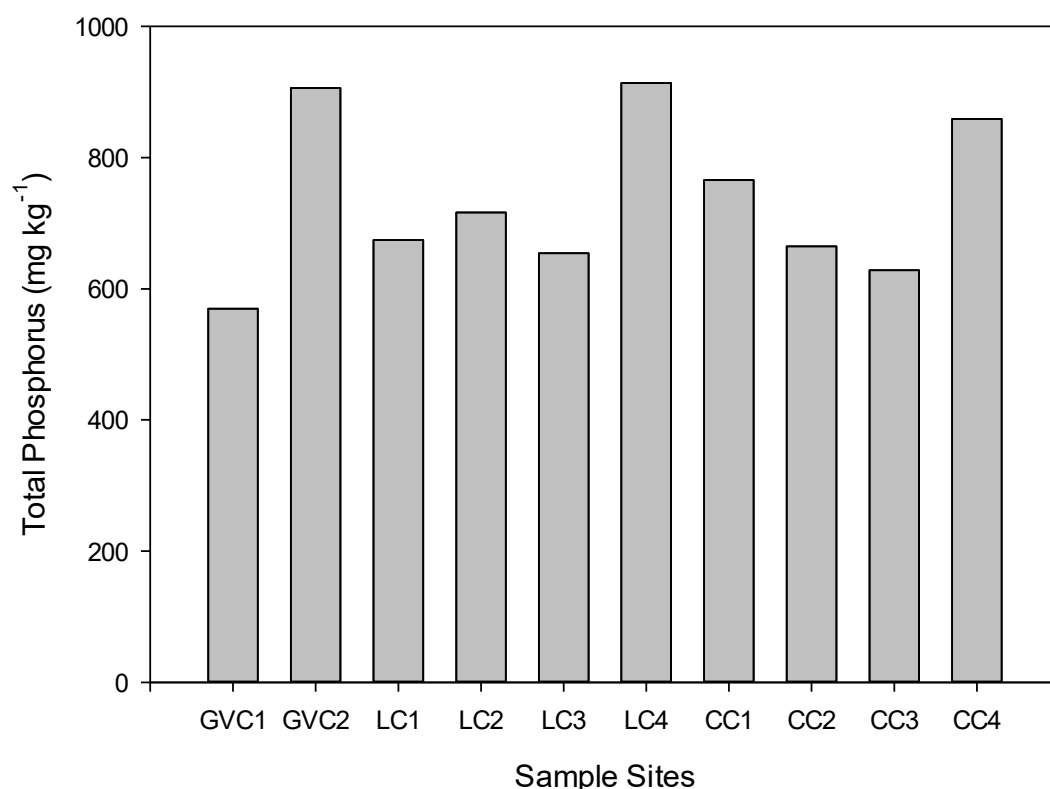


Figure 7: Total Phosphorus concentrations in the deposited sediment at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. Samples were taken during the second research trip and the data combined. Each bar represents the Total Phosphorus concentrations for a single sampling event.

4.3 Total Suspended Solids

The total suspended solids concentrations at CC4 varied significantly with all other sample sites (Figure 8) when analysed using a One-way ANOVA. CC4 was significantly different to GVC1 ($p=0.004$), GVC2 ($p=0.034$), LC1 ($p=0.026$) and LC4 ($p=0.015$). The highest concentration of total suspended solids were observed at CC4. Two-way ANOVA analysis did not show a significant difference in the concentration of total suspended solids over the different seasonal sampling dates when comparing the three catchments. There was a significant difference over the different seasonal sampling dates at sites within the Grandview Creek catchment ($p=0.001$). Significant seasonal differences between the different sites within the Lagoon Creek and

Cameron Creek catchments could not be tested for due to missing data not allowing the use of a Two Way ANOVA.

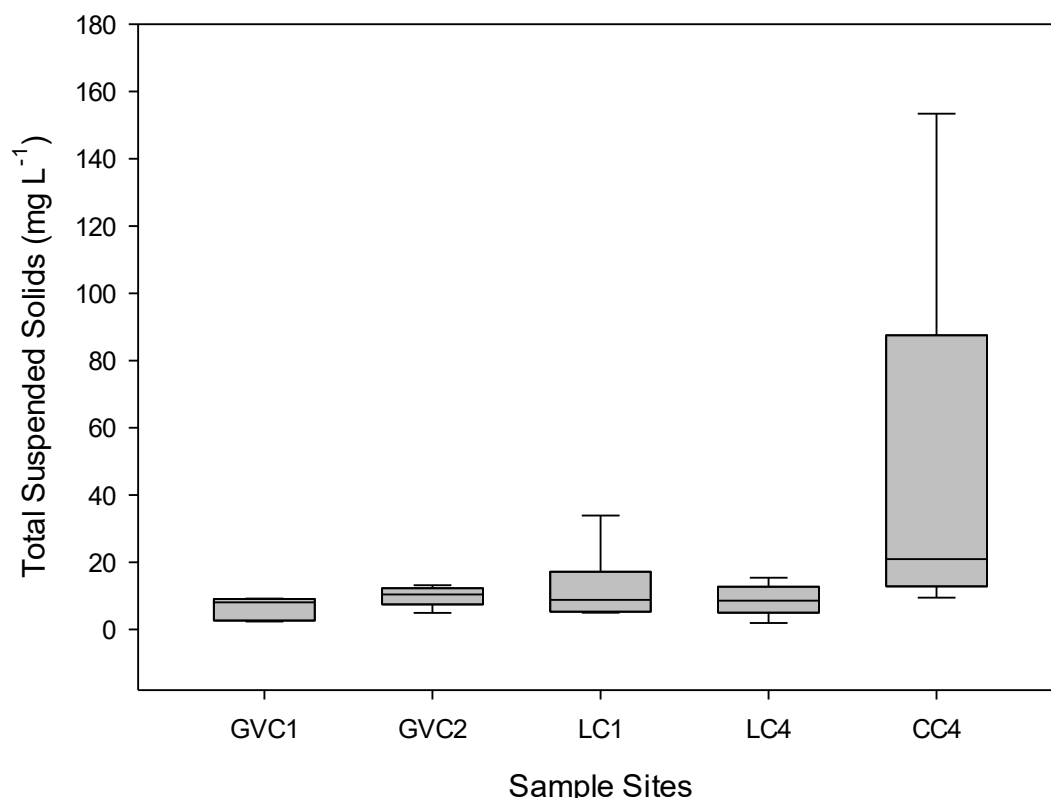


Figure 8: In-stream total suspended solids concentrations at each sample site, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with the median in the centre. The whiskers show the maximum and minimum values. Samples were collected over the duration of three research field trips and the data combined.

4.4 Visual Clarity

Visual clarity tube results varied significantly between sample sites (Figure 9) when analysed using a One-way ANOVA. The results from Cameron Creek site four varied significantly in comparison with Grandview Creek site one ($p=0.006$), Grandview Creek site two ($p=0.007$), Lagoon Creek site one ($p=0.007$), and Lagoon Creek site four ($p=0.007$). Lagoon Creek site two varied significantly in comparison with Grandview Creek site one ($p=0.021$), Grandview Creek site two ($p=0.025$), Lagoon Creek site one ($p=0.024$), and Lagoon Creek site four. Two-way ANOVA analysis showed a significant difference in visual clarity over the different seasonal sampling

dates when comparing the three catchments ($p=0.012$). There was no significant difference in visual clarity over the different seasonal sampling dates when comparing sites within the Grandview Creek, and Lagoon Creek catchments. A significant difference was observed over the different seasonal sampling dates at sites within the Cameron Creek catchment ($p= 0.010$).

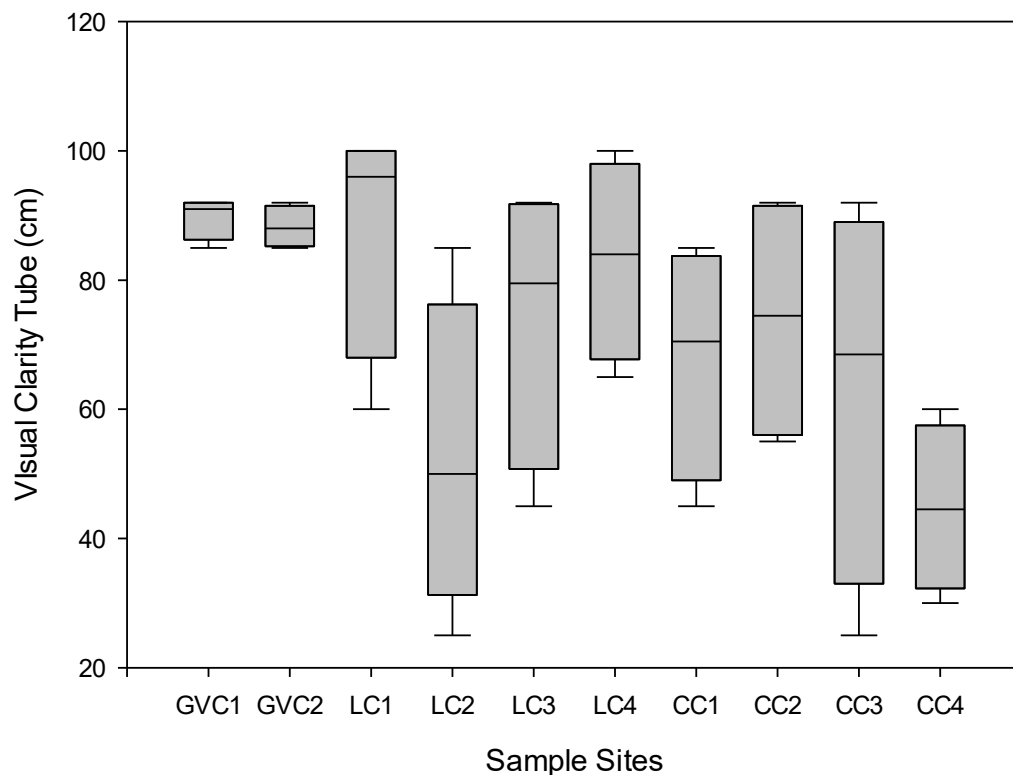


Figure 9: The visual clarity of the stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. Samples were collected over the duration of three research field trips and the data combined.

4.5 Conductivity

Conductivity levels varied significantly between sample sites (Figure 10) when analysed using a Kruskal-Wallis One Way Analysis of Variance. Lagoon Creek site two was significantly different to Grandview Creek site one ($h=36.1$, $p=0.004$) and Lagoon Creek site one ($h=36.1$, $p=0.001$), and Cameron Creek site one was significantly different to Grandview Creek site one ($h=36.1$, $p=0.036$) and Lagoon Creek site one ($h=36.1$, $p=0.010$). In each of the three catchments, the highest conductivity levels

came from the lowest elevation sample sites. The highest conductivity level was observed at Lagoon Creek site four. Significant differences over the different seasonal sampling dates were unable to be tested for using a Two-way ANOVA due to the non-normal distribution of the data, which was unable to be normally transformed.

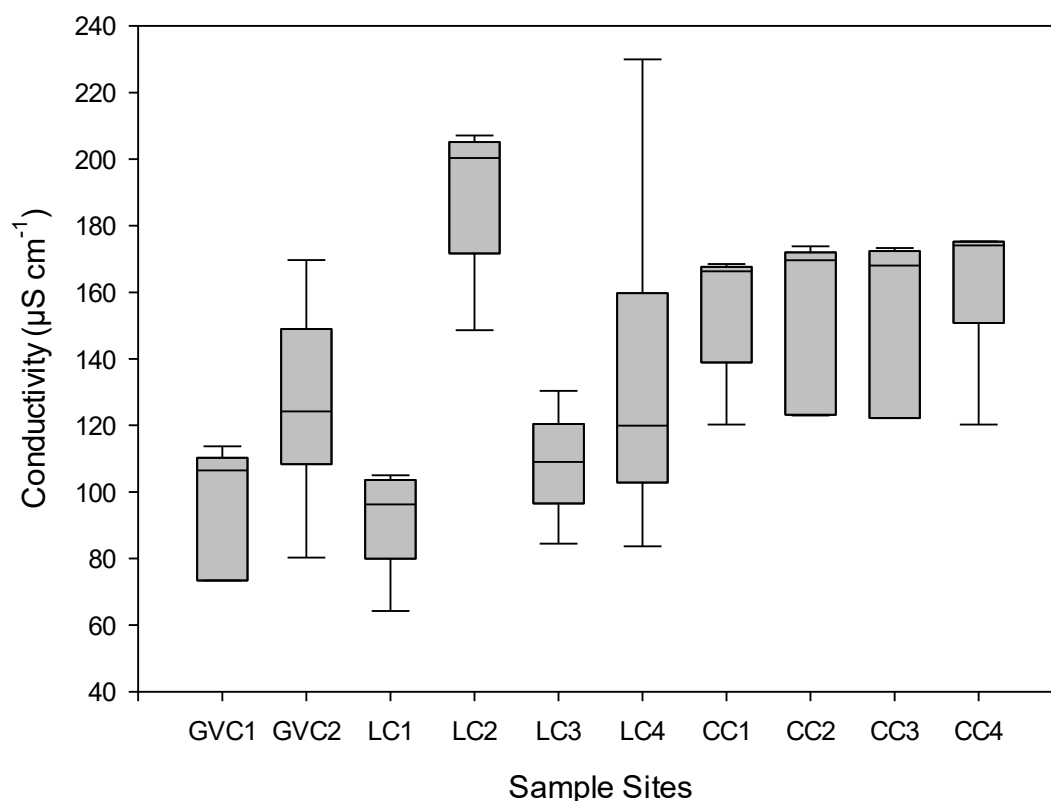


Figure 10: Conductivity levels of the stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. Conductivity was sampled over the duration of three research fieldtrips, and the data combined.

4.6 Dissolved Oxygen

The percentages of dissolved oxygen varied significantly between sample sites (Figure 11) when analysed using a One-way ANOVA. This variation was most notable at LC2, which was significantly different in comparison with GVC1 ($p=0.003$), GVC2 ($p<0.001$), LC1 ($p=0.008$) and CC2 ($p=0.010$). GVC1 was significantly different in comparison with CC3 ($p=0.047$) and CC4 ($p=0.025$). GVC2 was significantly different in comparison with LC3 ($p=0.008$), LC4 ($p=0.005$), CC1 ($p=0.009$), CC3 ($p=0.003$) and

CC4 ($p<0.001$). The lowest percentage of dissolved oxygen was observed at CC4. Two-way ANOVA analysis showed a significant difference in the percentages of dissolved oxygen over the different seasonal sampling dates when comparing the three catchments ($p=0.029$). Significant differences were observed over the different seasonal sampling dates at sites within the Grandview Creek catchment ($p=0.006$). No significant differences were observed at sites within the Lagoon Creek and Cameron Creek catchments.

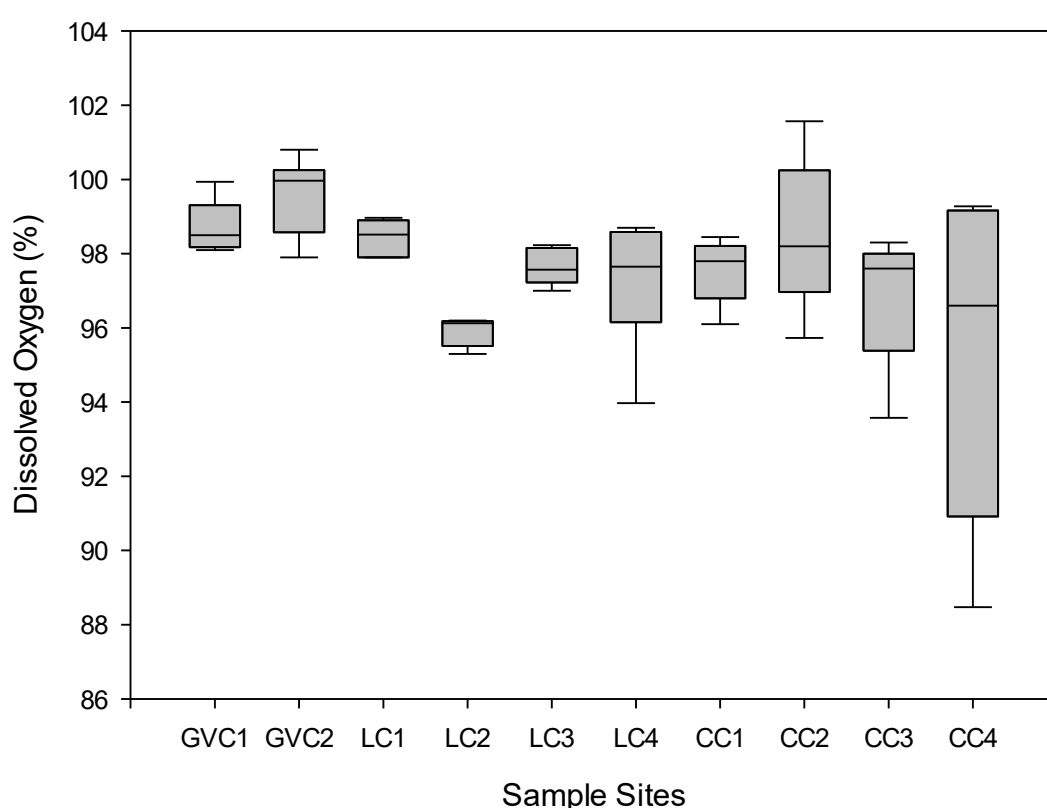


Figure 11: The percentage of dissolved oxygen at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. Dissolved oxygen was sampled over the duration of three research fieldtrips and the data combined.

4.7 pH

pH sample results did not show any significant differences between sample sites (Figure 12) when analysed using a One-way ANOVA. The largest variation in pH levels was observed at Grandview Creek site one. Two-way ANOVA analysis showed significant difference in pH levels over the different seasonal sampling dates when

comparing the three catchments ($p<0.001$). There was no significant differences observed over the different seasonal sampling dates at sites within the Grandview Creek catchment. There was a significant difference at sites within the Lagoon Creek ($p=0.037$), and the Cameron Creek catchments ($p=0.002$).

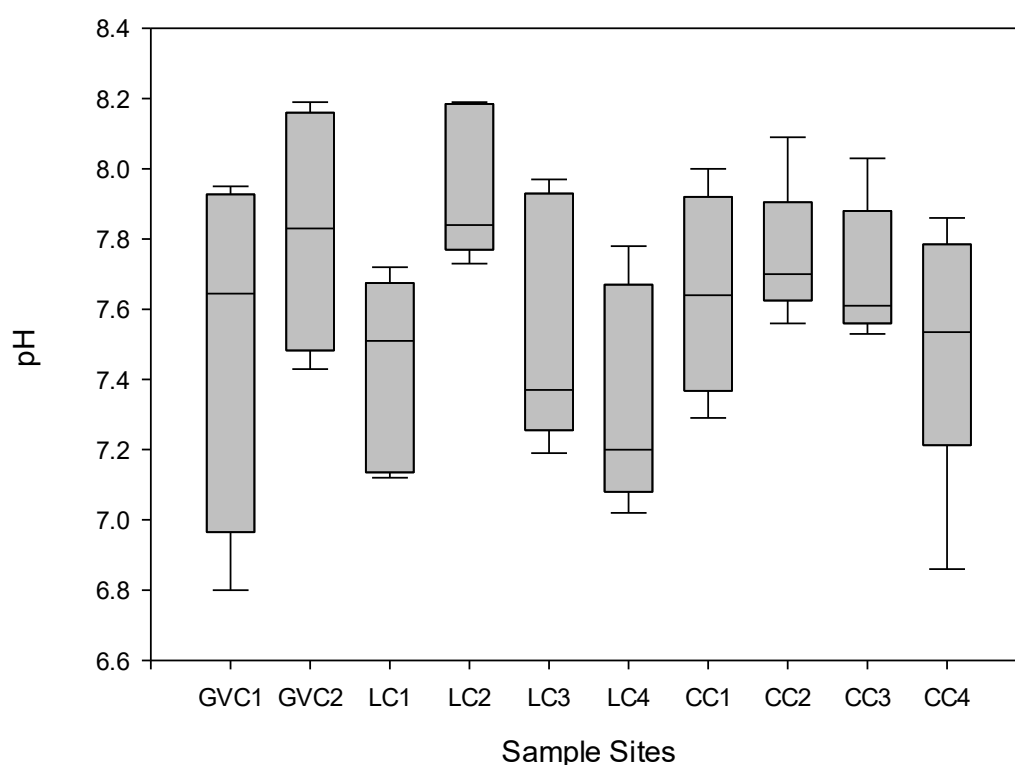


Figure 12: The pH levels in stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. pH was sampled over the duration of three research field trips and the data combined.

4.8 Stream Flow

Stream flow was not significantly different between sample sites (see Appendix E for box and whisker plot). The flow at LC2 was too low to be measured on any sample trip. Significant differences over the different seasonal sampling dates were unable to be tested for using a Two-way ANOVA due to the non-normal distribution of the data, which could not be normally transformed.

4.9 Stream Discharge

Stream discharge was not significantly different between sample sites (see Appendix F for box and whisker plot) when analysed using a One-way ANOVA. Two-way

ANOVA analysis did not show a significant difference in stream discharge over the different seasonal sampling dates when comparing the three catchments. There was no significant difference observed over the different seasonal sampling dates at sites within the Grandview Creek catchment. A significant interaction between date and stream discharge between sites in the Lagoon Creek and Cameron Creek catchments could not be tested for due to missing data not allowing the use of a Two-way ANOVA.

4.10 Correlation Analysis

This section provides the results of the correlation analyses that indicate some potential relationships between pairs of variables

4.10.1 Total Phosphorus vs. Total Dissolved Phosphorus

Plotting the results of Total Phosphorus against Total Dissolved Phosphorus indicated a positive relationship between the two variables, with a correlation coefficient of 0.67 (Figure 13).

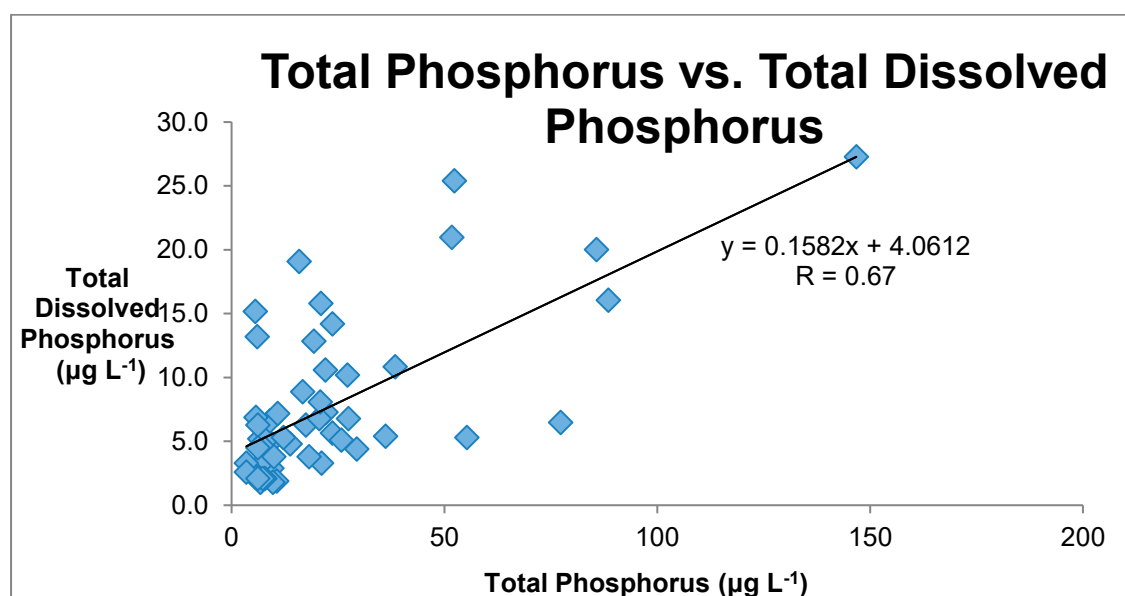


Figure 13: Total Phosphorus vs. Total Dissolved Phosphorus with a correlation coefficient (R) of 0.67.

4.10.2 Total Phosphorus vs. Diffusive Gradient in a Thin-film

Plotting the results of Total Phosphorus against DGT indicated a positive relationship between the two variables, with a correlation coefficient of 0.80 (Figure 14).

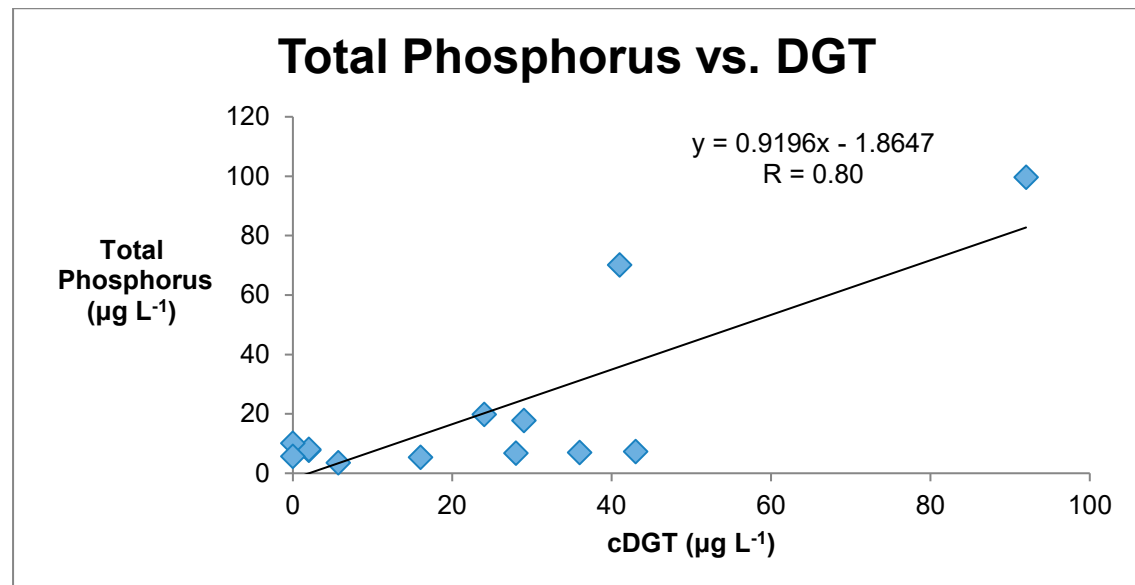


Figure 14: Total Phosphorus vs. DGT with a correlation coefficient (R) of 0.80.

4.10.3 Total Dissolved Phosphorus vs. Diffusive Gradient in a Thin-film

Plotting the results of Total Dissolved Phosphorus against DGT indicated a positive relationship, with a correlation coefficient of 0.79 (Figure 15).

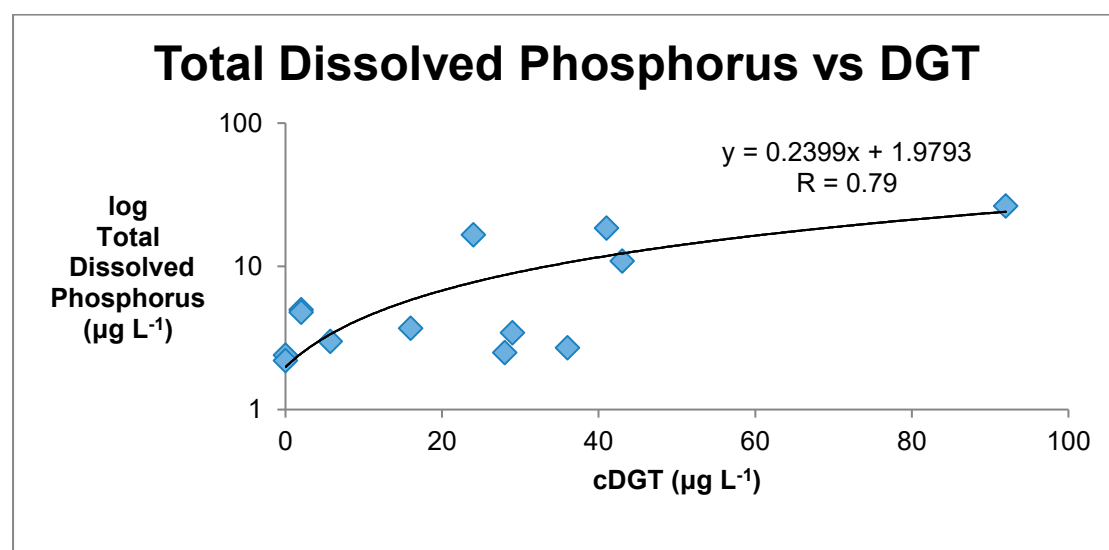


Figure 15: Total Dissolved Phosphorus vs. DGT with a correlation coefficient (R) of 0.79.

4.10.4. Total Phosphorus vs. Visual Clarity Tube

Plotting results of the visual clarity tube against Total Phosphorus indicated a negative relationship between the two variables, with a correlation coefficient of -0.74 (Figure 16).

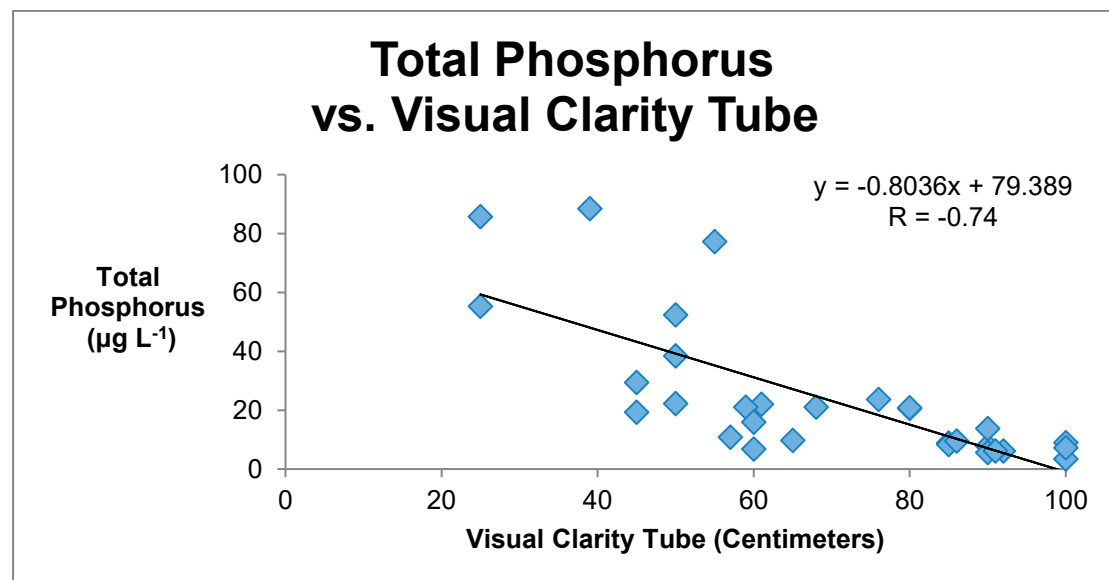


Figure 16: Visual clarity tube results plotted against Total Phosphorus with a correlation coefficient (R) of -0.74.

4.10.5 Total Phosphorus vs. Total Suspended Solids

Plotting results of Total Phosphorus against total suspended solids indicated a positive relationship between the two variables, with a correlation coefficient of 0.56 (Figure 17).

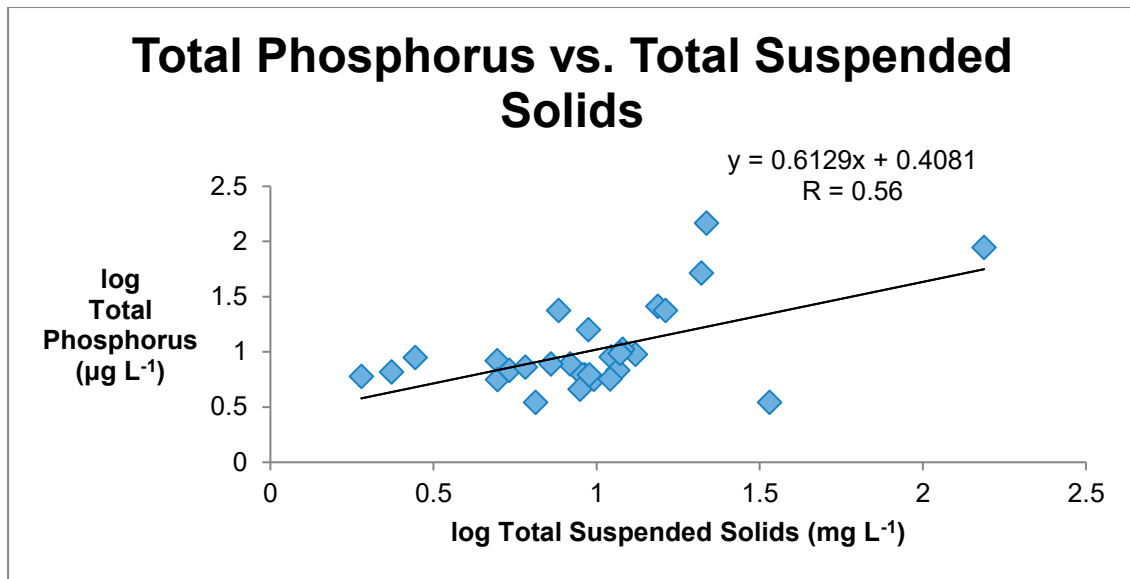


Figure 17: Total Phosphorus plotted against Total Suspended Solids with a correlation coefficient (R) of 0.56.

4.10.6 Total Phosphorus vs. Macroinvertebrate EPT Richness

Plotting the results of Total Phosphorus against macroinvertebrate EPT richness indicated a negative relationship between the two variables, with a correlation coefficient of -0.59 (Figure 18).

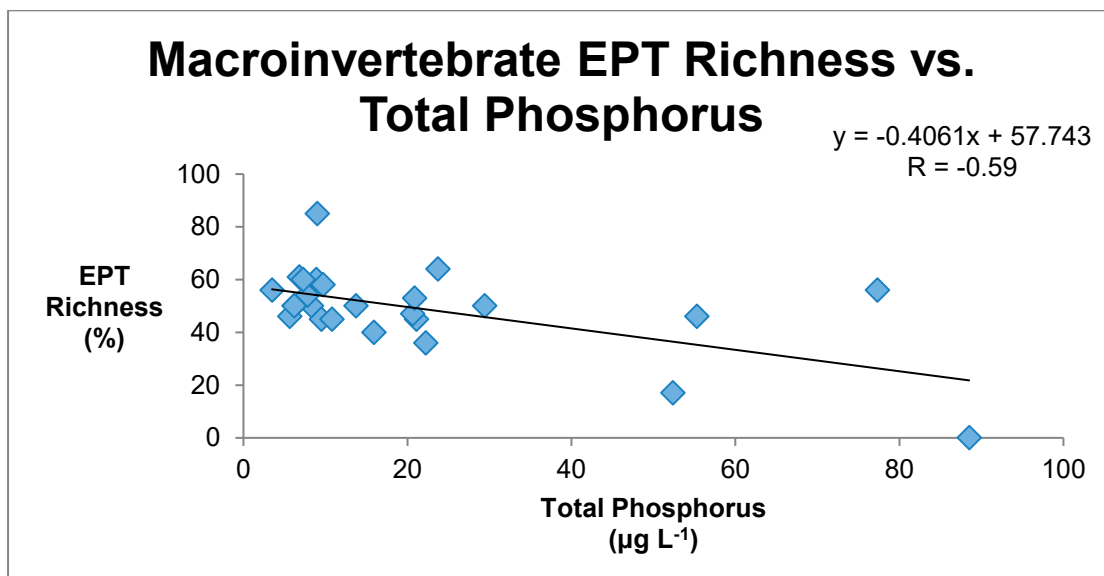


Figure 18: Macroinvertebrate EPT richness plotted against Total Phosphorus with a correlation coefficient (R) of -0.59.

4.11 Principle Component Analysis

Principle component analysis of the abiotic variables between catchments showed a difference in the ordination space occupied by the Cameron Creek catchment (Figure 19). The Cameron Creek catchment was weighted towards the positive end of Axis One, which is dominated by Total Phosphorus, Total Dissolved Phosphorus and visual clarity. Axis One explained 45% of the variance, whilst Axis Two explained 21%. Total suspended solids, DGT, phosphorus in deposited sediment and phosphorus in riparian soils were excluded due the difference in sampling repetition resulting in missing values. A separate analysis was undertaken for the macroinvertebrates due to the difference in sampling repetition.

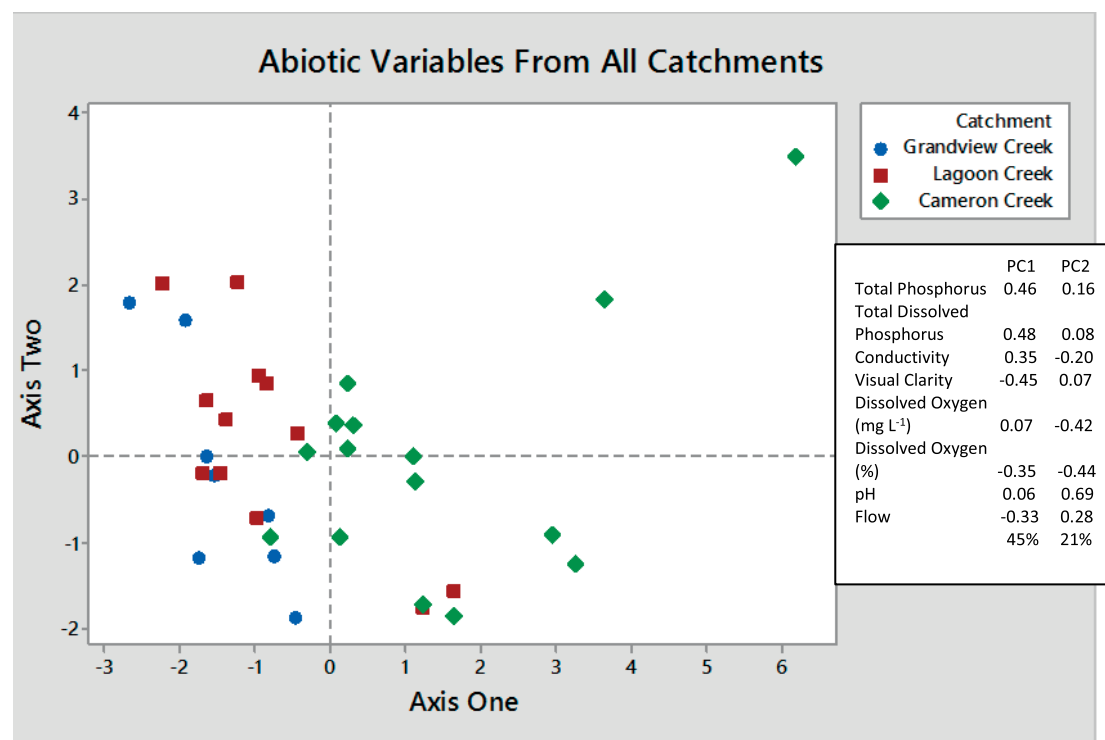


Figure 19: Principal Component Analysis of monitored abiotic variables. The points represent each sample location in each catchment and the relative loadings are tabulated adjacent to the figure.

Principle component analysis of macroinvertebrates between catchments revealed a difference in the ordination space occupied by the Cameron Creek catchment (Figure 20). The Cameron Creek catchment was weighted towards the negative end of Axis One, which is dominated by MCI scores and EPT taxa richness percentages. Axis One explained 77% of the variation, whilst Axis Two explained 23%.

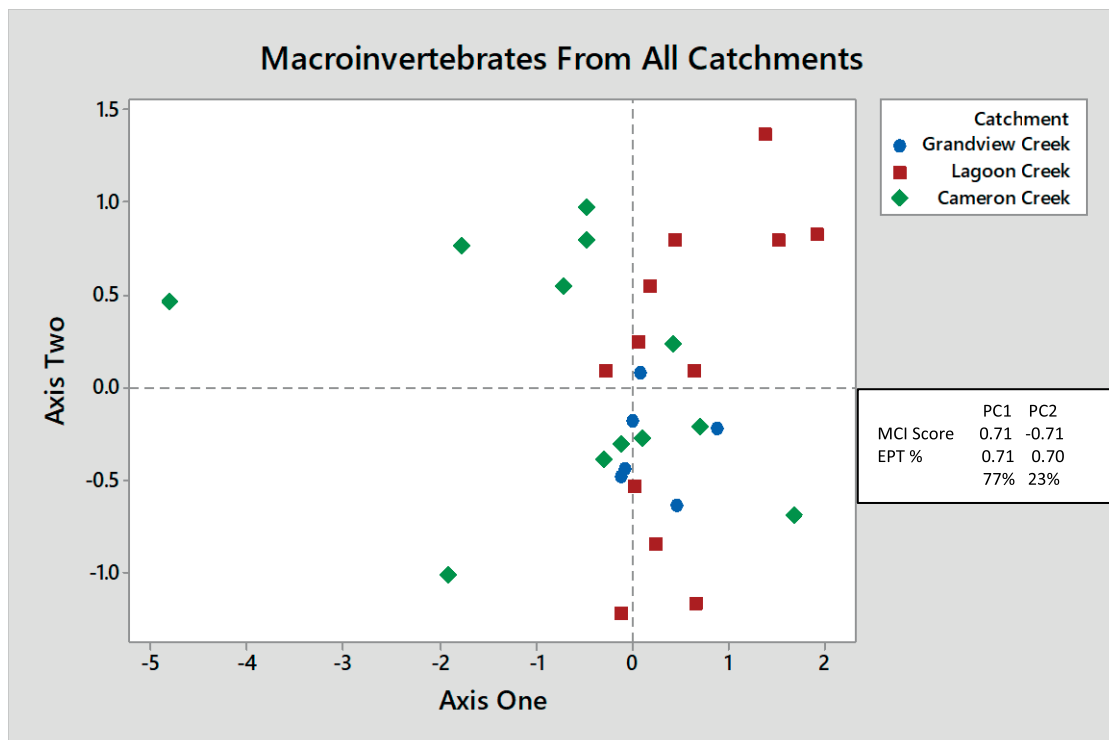


Figure 20: Principal Component Analysis of the macroinvertebrate data. The points represent each sample location in each catchment and the relative loadings are tabulated adjacent to the figure.

Chapter Five

Discussion

5.1 Macroinvertebrates

The lowest MCI score (46) and EPT taxa percentages (0%) were recorded at CC4. MCI scores <99 are indicative of a fair quality water class with probable moderate pollution, and scores of <80 are indicative of a poor water quality class with probable severe pollution (Stark and Maxted 2007). For EPT taxa, percentages <51 are indicative of fair water quality with probable moderate pollution, and percentages <25 are indicative of poor water quality with probable severe pollution (LAWA 2013).

MCI scores from the Grandview Creek catchment all fall within the good water quality class, indicative of possible mild pollution. However, all EPT taxa richness sample results from GVC2, and one sample result from GVC1 fall within the fair water quality class. Although many of the EPT taxa percentages at sample sites in the Grandview Creek catchment are classed as fair, these results are near the upper threshold. In the Lagoon Creek catchment, all MCI scores were again indicative of a good quality class of water. However, at Lagoon Creek Site Two (LC2) and Lagoon Creek Site Three (LC3), two EPT taxa percentage sample results from each of these sites fall within the fair water quality class, indicative of probable moderate pollution. The lowest of these was 36%, recorded at Lagoon Creek Site Two. The other three fair quality EPT taxa percentage results in the Lagoon Creek catchment were all near the threshold of the quality class. In the Cameron Creek catchment, several sample results fall within the poor water quality class for both MCI scores and EPT taxa percentages, indicative of probable severe pollution. Of most concern in the Cameron Creek catchment is CC4, where sample results falling in the poor water quality class for both MCI scores and EPT taxa richness were regularly recorded.

The lower MCI scores and EPT taxa richness percentages recorded at the bottom of the Cameron Creek catchment (where there are no areas of significant native

vegetation, minimal riparian vegetation, and the stream is running through cultivated land) are consistent with other findings in the literature. These findings document a consistent reduction in macroinvertebrate taxonomic richness in streams that are draining from agricultural land, in comparison to those situated in less developed catchments (e.g. Baumgartner and Robinson 2017; Doledec et al. 2011; Fierro et al. 2017; Genito et al. 2002; Harding and Winterbourn 1995; Larned et al. 2016; Quinn and Hickey 1990). Nutrient concentrations were also highest at the bottom of the Cameron Creek catchment, and the findings of the current research project are similar to that of another study undertaken in the Otago Region, which focused on 12 streams along a gradient of agricultural development (Niyogi et al. 2003). This study demonstrated that lower MCI scores were occurring with higher concentrations of nutrients.

Like the results recorded at CC4, findings in the literature also document that intensified land-use causes not only an overall reduction in taxonomic richness, but also the replacement of EPT taxa with those that are more tolerant to pollution (e.g. Lenat and Crawford 1994; Ramezani et al. 2016; Wallace et al. 1996; Wright-Stow and Wilcock 2017). The results recorded at CC4 show a clear reduction in overall taxonomic richness in comparison with other sample sites, and low EPT taxa richness percentages that are indicative of probable moderate or severe pollution (see Appendix C for the total abundance of macroinvertebrate individuals from each sampling event). The thick layer of deposited sediment observed at CC4 is likely to be an important factor in the reduction of macroinvertebrate habitat, as sediment deposits are known to reduce their diversity (Suren et al. 2005), and the quality of their habitat (Quinn and Hickey 1990).

The lack of notable native vegetation in the Cameron Creek catchment, and the minimal riparian vegetation at lower elevations is likely to have resulted in a reduction of macroinvertebrate food supply. The lack of vegetation has also likely helped to facilitate the mobilisation of increased concentrations of phosphorus bound to sedimentation to the waterway.

No seasonal variation was observed when comparing the three catchments for either MCI scores or the percentage of EPT taxa. The winter season in which this

research project was undertaken was very mild (R.McNeilly, personal communication, September 18, 2016). Had the winter months been more typical of the winter climate at Mt. Grand Station during this research project, more significant seasonal differences in the freshwater macroinvertebrate populations may have been recorded.

5.2 Phosphorus

With the exception of LC2, Total Phosphorus results from each of the three catchments saw concentrations increase sequentially from site to site at lower elevations down-stream. These results are consistent with findings in the literature, where contaminate concentrations are often compounded further down-stream as tributary flows accumulate (Larned et al. 2004). Each of the streams of focus at Mt. Grand Station in the present research are fed by multiple smaller tributary waterways that drain from various intensities of over-sown and top-dressed hillside. A Total Phosphorus trigger value for New Zealand upland streams and rivers that are non-glacial or lake fed has been set at $24 \mu\text{g L}^{-1}$. Concentrations of Total Phosphorus recorded above $24 \mu\text{g L}^{-1}$ are intended to trigger a management response (Davies-Colley 2000). The Grandview Creek catchment recorded a maximum concentration of Total Phosphorus of $10.6 \mu\text{g L}^{-1}$, indicating that no management response is required. The maximum concentration recorded in the Lagoon Creek catchment was $55 \mu\text{g L}^{-1}$, taken at LC2 where stream discharge was very low. However, the maximum concentration recorded at the bottom of the Lagoon Creek catchment at LC4 was $26 \mu\text{g L}^{-1}$. The results from LC2 indicate that a management response is required. However, further down-stream at the higher discharging LC4, the maximum Total Phosphorus concentration falls slightly above the trigger value. This indicates that LC2 may be a significant tributary source of Total Phosphorus in the Lagoon Creek catchment. However, the low discharge levels will likely lead to heightened $\mu\text{g L}^{-1}$ concentrations when phosphorus enters the waterway at this sample site. In the Cameron Creek catchment the maximum concentration of Total Phosphorus recorded was $146.8 \mu\text{g L}^{-1}$. This concentration was recorded at CC4, where 83% of samples exceeded the Total Phosphorus trigger value. CC4 is at the

bottom of the very steep Cameron Creek catchment, and the Total Phosphorus results indicate that in comparison to the other two catchments, large concentrations of phosphorus bound to soil particles are being mobilised to the waterway. The high Total Phosphorus concentrations recorded indicate that a management response is required for the lower altitudes of the Cameron Creek catchment.

Total Dissolved Phosphorus results from sample sites in the Lagoon Creek and Cameron Creek catchments displayed some variation, whilst the Grandview Creek catchment results were more consistent, recording the lowest maximum concentration of $5 \mu\text{g L}^{-1}$. The Total Dissolved Phosphorus results also indicate that ecological water quality is not a concern in the Grandview Creek catchment. However there may be some concerns in the Lagoon Creek and Cameron Creek catchments. The Lagoon Creek catchment recorded a maximum concentration of $15 \mu\text{g L}^{-1}$. The maximum concentration in the Cameron Creek catchment was $27 \mu\text{g L}^{-1}$. At the bottom of the Cameron Creek catchment, the minimum concentration recorded was $14 \mu\text{g L}^{-1}$. Schedule 15 of the Otago Regional Plan has set a Dissolved Reactive Phosphorus target of $5 \mu\text{g L}^{-1}$, with Dissolved Reactive Phosphorus forming a smaller component of Total Dissolved Phosphorus. The higher concentrations of Total Dissolved Phosphorus recorded in the Lagoon Creek and Cameron Creek catchments indicate that on-going monitoring is required to ensure Mt. Grand Station meets their responsibilities set out under Schedule 15.

Although the cDGT phosphorus results were not significantly different between sample sites, the highest maximum concentration of $92 \mu\text{g L}^{-1}$ was recorded at CC4. GVC2 recorded the lowest maximum concentration, where two results were below the $1.5 \mu\text{g L}^{-1}$ method detection level and a concentration of $2 \mu\text{g L}^{-1}$ was recorded. The cDGT results reinforce the Total Phosphorus and Total Dissolved Phosphorus results, showing that the bottom of the Cameron Creek catchment is of most concern in regards to increased phosphorus concentrations in the waterways of Mt. Grand Station.

The highest concentration of Total Phosphorus in riparian soil was recorded at CC4, with a concentration of 1132 mg kg^{-1} . In the Grandview Creek and Lagoon Creek

catchments, each containing two riparian soils sample sites, Total Phosphorus concentrations decreased sequentially at lower elevations down-stream. The lowest concentration of 378 mg kg^{-1} of Total Phosphorus in riparian soils was recorded at Lagoon Creek site four. The higher concentrations recorded at CC4 are likely to be a result of the area of nearby pasture. Interestingly the concentrations reduce in elevation for both the Grandview Creek and Lagoon Creek catchments. Sites at higher-elevations in both of these catchments are surrounded by increased native vegetation in comparison with sample sites at lower elevations, and there is no clear reason as to why the Total Phosphorus concentrations of riparian soils are decreasing down-stream. The phosphorus concentrations in riparian soils were only sampled at CC4 in the Cameron Creek catchment so a comparison between high and low elevations could not be made.

For each of the three catchments, the highest concentration of Total Phosphorus in the stream sediment was observed at the sample site of lowest elevation. The highest concentration was observed at Lagoon Creek site four, which was 913.7 mg kg^{-1} . The lowest concentration of 569.6 mg kg^{-1} was recorded at Grandview Creek site one. A comparison of the stream sediment Total Phosphorus concentrations results with those of the riparian soils (excluding Cameron Creek catchment which had one riparian soil sample site) indicates that the majority of the stream sediment is being sourced from up-stream in the catchment, where there are higher phosphorus riparian soil concentrations. The lowest Total Phosphorus concentrations in riparian soils were observed at low-elevations, in comparison with the highest Total Phosphorus concentrations of deposited sediment that were also observed at low-elevations. This indicates that instream sediment is being mobilised from higher elevations in the catchment, where the Total Phosphorus content of the riparian soils is the highest, before being deposited out of suspension at lower elevations as the flow rate decreases.

With the exception of the sediment sample results which were highest at LC4, the highest concentrations for all other phosphate measurements were recorded at CC4. These samples were taken at the bottom of the Cameron Creek catchment where there are no notable areas of native vegetation, minimal riparian vegetation,

no conservation area, and the stream is running through cultivated land. These results are consistent with those in the literature, which have shown that the vegetation structure and land-use intensity within a catchment will have a strong influence on stream nutrient concentrations (Ballantine and Davies-Colley 2014; Quinn and Hickey 1990; Smith et al. 1993).

LC1 and LC3 are similar in elevation but drain from two different sub-catchments of Lagoon Creek. Total Phosphorus concentrations were higher at LC3 (with no significant native vegetation on the stream banks in the catchment above) in comparison with LC1 (with an area of kānuka forest on the stream banks in the catchment above). All but one Total Dissolved Phosphorus sample collected from LC3 were higher in concentration than those collected from LC1. The lower phosphorus concentration results recorded at LC1 in comparison with LC3 indicate that the area of native vegetation on the stream banks above the sample site may be having an effect on phosphorus concentrations in the stream water. LC2 recorded higher phosphorus concentrations than LC1 and LC3, however stream discharge was much lower at this site. LC2 is draining from an area of established kānuka forest. However, there are also very steep areas on either side of the sample site that are dominated by pasture grass which is over-hanging much of the up-stream margins of LC2. The low discharge level, steep terrain and close proximity of pasture grass are likely to be responsible for the increased phosphorus concentrations recorded at LC2. Conversion of native forest to pasture grass is known to cause profound changes in stream health (Hicks 1997; Verheyen et al. 2015). The very low discharge levels at LC2 will likely result in increased $\mu\text{g L}^{-1}$ concentrations of phosphorus as runoff enters the stream, in comparison with higher discharging sites that contain more water to dilute the $\mu\text{g L}^{-1}$ concentrations. The low discharge also explains the increased conductivity levels recorded at LC2.

There was no significant seasonal variation observed between catchments in Total Phosphorus and cDGT concentrations. However a significant seasonal variation was observed for Total Dissolved Phosphorus. This suggests that the different forms of phosphorus may be being mobilised to the waterways under different environmental conditions. If the sampling for this research project had included a

harsher winter, and had captured a significant rainfall event, then greater seasonal variation of phosphorus concentrations may have been observed (see Appendix G for rainfall data during research project).

5.3 Total Suspended Solids

The highest maximum concentration of total suspended solids was recorded at CC4, which was significantly higher than all other sample sites. At CC4 the stream is running through an area of cultivated land with minimal riparian vegetation, and there are no areas of notable native vegetation higher in the catchment. The higher concentrations of total suspended solids recorded at CC4 are consistent with the findings in the literature, where intensified agriculture has been identified as a major source of sediment for New Zealand's freshwater resources (Smith et al. 1993; Vant 1999). The results indicate that the adjacent developed pastureland upstream of CC4 is facilitating the mobilisation of increased suspended solids particles to the stream.

Catchment vegetation structure is also known to influence stream sediment inputs (Miller et al. 2011), along with the immediate riparian vegetation (Fennessy and Cronk 1997; Lam et al. 2011). The high total suspended solids concentrations recorded at CC4 indicate that the lack of notable vegetation along the riparian margins, and throughout the wider catchment, may be facilitating the mobilisation of sediment from land into the stream. Although it was not quantified, thick layers of deposited sediment were also observed at CC4 during all research trips. Deposited sediment was observed in all three catchments, however on each research trip it was most prominent at CC4. The deposits indicate that sediment is being mobilised to the streams in high precipitation events under increased discharge (particularly in the Cameron Creek catchment), and is then deposited from suspension when the discharge declines. Visual clarity and total suspended solids data show that large quantities of sediment were not constantly moving down-stream in the waterways whilst sampling was undertaken.

No significant seasonal differences were observed between the three catchments. Rainfall levels during each of the research trips were low, with 1.4mm recorded during December and none for the other two sampling events. Had a larger

rainfall event occurred whilst sampling was being undertaken, then greater seasonal variation for total suspended solids concentrations may have been observed.

5.4 Visual Clarity

LC2 recorded the lowest visual clarity result, however the discharge at this sample site was very low during all research trips and it was hard to fill the visual clarity tube without disturbing the stream substrate. Visual clarity was consistently low at the bottom of the Cameron Creek catchment where there is minimal riparian vegetation and no notable areas of native vegetation or conservation area. The visual clarity at the bottom of the Cameron Creek catchment was significantly reduced in comparison with sites at the bottom of the Grandview Creek and Lagoon Creek catchments. This indicates that the areas of significant native vegetation in these two catchments may be helping to facilitate the increased clarity of the waterways in comparison with the bottom of the Cameron Creek catchment.

A significant difference in seasonal variation was observed between the three catchments, and at sites within the Cameron Creek catchment. Cameron Creek runs through Mt. Grand Station's area of developed pasture, with the associated sediment from this area then easily mobilised to the waterway. This may explain why the seasonal differences in visual clarity were significantly noticeable within the Cameron Creek catchment and not the others.

5.5 Dissolved Oxygen

Much of the dissolved oxygen percent saturation results fall below trigger guideline thresholds for New Zealand upland rivers, which are 99-103%. However, it is noted that dissolved oxygen may not be very useful as a trigger value due to diurnal and seasonal variation (ANZECC and ARMCANZ 2000). The greatest variation in dissolved oxygen percentages was observed in the Cameron Creek catchment. Of all sample sites in the three different catchments, CC2 recorded the highest result of 101.57%, and CC4 recorded the lowest result of 88.47%. All but two sample results from CC4 fall below the trigger guidelines and may be indicative of water quality issues.

5.6 Correlations

Total Phosphorus results show a positive relationship with the results of Total Dissolved Phosphorus, predicting 45% of their variability. Total Phosphorus results also show a positive relationship with the DGT results, predicting 65% of their variability. Total Dissolved Phosphorus results show a positive relationship with the DGT results, predicting 62% of their variability. Although the various phosphorus sample results relate positively to each other, and are able to explain some of the variability for the dependant variable, in each of these correlations there is still variability that cannot be explained. The correlated phosphorus results highlight the benefit of sampling for various forms of phosphate, in order to gain a more thorough understanding of how each form may be entering a stream. This is due to the unexplained variability in the phosphorus correlation analyses. For example, correlation results show that the monitoring Total Phosphorus can enable the prediction of 45% of Total Dissolved Phosphorus variability, but to gain a more accurate understanding both phosphates should be monitored.

Results from the visual clarity tube showed a negative relationship with the results of Total Phosphorus, predicting 55% of their variability. This correlation is demonstrating that sediment bound with Total Phosphorus particles contribute to the reduction in visual clarity. Although much of the Total Phosphorus variability was not able to be predicted by the visual clarity tube, this correlation demonstrates that the clarity tube can still be a useful monitoring tool when monitoring for ecological water quality issues. The visual clarity tube is simple to use and can assist land-users in predicting where an increase of Total Phosphorus may be occurring, before using this information to develop a more thorough stream monitoring plan for implementation.

A positive relationship was demonstrated between the results of total suspended solids and the results of Total Phosphorus. The results of total suspended solids were able to predict 32% of the variability within the results of Total Phosphorus. This relationship indicates that a large component of the total

suspended solids samples were comprised of other organic matter and not dominated by suspended sediment particles bound with Total Phosphorus.

Total Phosphorus results showed a negative relationship with macroinvertebrate EPT taxa percentages, explaining 35% of their variability. This relationship demonstrates that although Total Phosphorus can explain some of the variability within the EPT taxa percentages, it is likely that interactions between multiple variables will determine the macroinvertebrate habitat suitability of a stream sites.

5.7 Principle Component Analysis

The Principle Component Analysis for the abiotic variables, and macroinvertebrates from all catchments showed that the Cameron Creek catchment largely occupies its own ordination space. The Cameron Creek catchment dominates the positive end for phosphorus and sediment concentrations, and the negative end for macroinvertebrate biodiversity. The separation of the Cameron Creek catchment in the Principle Component Analysis fits the general trend of the other results, where phosphate concentrations are increased and macroinvertebrate biodiversity is reduced in the lower elevations of Cameron Creek, and indicates that this is where any management efforts would best be implemented.

Chapter Six

Conclusions and Future Work

6.1 Conclusions

The results of this research project have demonstrated that sedimentation and phosphate concentrations in the waterways of Mt. Grand Station significantly vary in relation to catchment vegetation structure. Visual clarity and total suspended solids results have shown that sedimentation was highest at the bottom of the Cameron Creek catchment (CC4), which contains no notable areas of native vegetation on either the lower and middle altitude slopes; there is also minimal riparian vegetation and the stream is running through cultivated land. Although not quantified, there was a thick layer of deposited sediment observed at the bottom of the Cameron Creek catchment on all research trips. There was deposited sediment observed at other sample sites, but this was by far the most prominent at CC4. Phosphate concentrations were also consistently highest at the bottom of the Cameron Creek catchment. This research project has demonstrated that a reduction in native vegetation cover, and an area of developed pasture are both assisting to increase the mobilisation of sediment and phosphorus concentrations to the waterway at the bottom of the Cameron Creek catchment.

The findings have also demonstrated that increased concentrations of suspended sediment and phosphorus are having a negative effect on benthic macroinvertebrate biodiversity at Mt. Grand Station. MCI scores and EPT taxa richness percentages were consistently lowest at the bottom of the Cameron Creek catchment, where increased sediment and phosphorus concentrations have been mobilised to the waterway. This has likely resulted in a reduction in macroinvertebrate habitat. The lack of native vegetation in either the lower and middle altitude slopes of the Cameron Creek catchment suggests that sediment and phosphorus concentrations, as well as catchment vegetation, are important factors in determining the habitat suitability for benthic macroinvertebrates.

In the context of managing New Zealand High Country waterways, an inference of this research project is that establishing a native riparian buffer zone along the lower altitude sections of the Cameron Creek catchment may help to prevent phosphorus from reaching the waterway, and assist Mt. Grand Station in meeting their requirements under Schedule 15 of the Otago Regional Plan. Monitoring for the relevant requirements set out under Schedule 15 should be undertaken over a rolling five-year period, when flows are at or below median flow. The thresholds are achieved once 80% of the samples meet, or are better than, the limits set out in Schedule 15. If any problem areas are identified by this five-year monitoring programme, the regional council request mitigation efforts (such as riparian restoration) are put in place so that the ecological health of the waterway can improve, and the relevant requirements are achieved. The Otago Regional Council is requesting that land-users begin monitoring as soon as possible, to assist them in meeting their Schedule 15 requirements by 2025.

The Otago Regional Council wishes to work with land-users as opposed to taking an immediate punitive approach. If on-going monitoring does demonstrate a water quality issue in regards to ecological health, the implementation of best management practises will go a long way to satisfying them that a land-user is serious about trying to mitigate their negative effects and improve water quality on their property. In the event that on-going monitoring is still consistently showing that stream water quality is unlikely to meet the Schedule 15 target guidelines, Lincoln University can request the Otago Regional Council's assistance in identifying where and how a particular waterway at Mt. Grand Station is being degraded. This is part of the regional council's responsibilities set out under Schedule 16 of the regional plan (Otago Regional Council Policy Advisor, personal communication, November 21, 2017).

6.2 Research Limitations

The conclusions of this research are limited by various factors. Research trips were limited to once per season over three seasons, and the number of sampling locations

on each trip was limited by the time, resources and vehicle space available, and vehicle space also limited the amount of samples that could be taken on each research trip. The availability of time and resources were also limitations on accurately quantifying catchment vegetation. When planning for this research project, it was intended that time integrated sediment devices would be used to gain a more thorough understanding of how much suspended sediment was moving through each catchment over extended sampling periods (Phillips et al. 2000). However, these devices proved to be too long, and were unsuitable for the higher altitude streams of Mt. Grand Station.

Increasing the frequency and duration of research trips would assist this research project to provide more detailed information on seasonal variation and allow for samples to be taken during rainfall events. Although the conclusions are limited by the frequency and duration of research trips, the macroinvertebrate communities and phosphorus soil concentrations present at each sample site reflect environmental conditions over a longer time-scale.

Limitations on time, resources and vehicle space meant that only a small number of sites could be sampled in each catchment. To truly achieve the goal of linking in-stream macroinvertebrates, physico-chemical conditions and land-use at specific sample sites would have required a different study design, comprising of a higher number of sample sites along a quantified gradient of catchment land-use intensity.

6.3 Future Work

Although the results of this research project highlight the benefits of sampling waterways for their concentrations of Total Phosphorus, Total Reactive Phosphorus and cDGT concentrations to gain more accurate information on how various forms of phosphorus are being mobilised to the waterways, if a recommendation was to be made as to which phosphorus methods should be prioritised, Total Dissolved Phosphorus (although ideally Dissolved Reactive Phosphorus should be sampled for) accompanied by cDGT would be recommended. These methods are prioritised

because Reactive Phosphorus has greater relevance than Total Phosphorus in the context of New Zealand stream management and the relevant statutory legislation.

In addition to prioritising Dissolved Reactive Phosphorus or Total Dissolved Phosphorus, future work monitoring the waterways of Mt. Grand Station should also sample the other parameters that are required to be monitored under Schedule 15 of the Otago Regional Plan (Otago Regional Council 2015). The sampling of Dissolved Reactive Phosphorus will require more prompt analysis than what was capable for this research project in order avoid potential phosphorus speciation.

The frequency and duration of research trips should be increased so that sampling can better capture seasonal variation and rainfall events, and meet the monitoring criteria set out in Schedule 15. High rainfall events are thought to be the primary controlling mechanism of phosphorus transport for streams of higher elevation (Caruso 2000). It is also recommended that additional sites should be sampled for their phosphorus concentrations in the LC2 sub-catchment, in order to gain a better understanding of how phosphorus is being mobilised to the waterway in this area. Future work should also aim to accurately quantify the vegetation structure of each catchment, which was beyond the scope of this research project, due to time restrictions and human and technical resources in the field.

The application of Superphosphate in the Cameron Creek catchment was scheduled for the summer after the completion of fieldwork for this research project. Superphosphate application will likely result in a further increase of phosphorus concentrations being mobilised to the waterway, and future work should monitor the phosphorus concentrations of Cameron Creek after fertiliser application. Time integrated sediment devices that are more suitable to high country environments that can be left in steep, high altitude streams should also be designed.

This research project has identified areas of Mt. Grand Station's waterways where ecological health may be a potential issue, and future monitoring is required to ensure that the station meets its requirements set out in Schedule 15 of the Otago Regional Plan.

Appendix A

Contaminant concentration limits for Receiving Water Group 3 that are applicable to Mt. Grand Station, taken from Schedule 15 of the Otago Regional Plan (Otago Regional Council 2015).

	Nitrate-nitrite nitrogen	Dissolved reactive phosphorus	Ammoniacal nitrogen	<i>Escherichia coli</i>	Turbidity
	0.075 mg/l	0.005 mg/l	0.01 mg/l	50 cfu/100 ml	3 NTU
Clutha/Mata-Au , above Luggate	31 March 2012				
Dart	31 March 2012	31 March 2012	31 March 2012	31 March 2012	Exempt
Kawarau , upstream of the Shotover confluence	31 March 2012				
Matukituki	31 March 2012	31 March 2012	31 March 2012	31 March 2012	Exempt
Tributaries to Lakes Hawea, Wakatipu, & Wanaka	31 March 2012				

The limits for Groups 4 and 5 are achieved when 80% of samples collected at a site, over a rolling 5-year period, meet or are better than the limits in Schedule 15.

A target date of 31 March 2025 is set when the contaminant concentration does not meet the limit as at 31 March 2012.

Appendix B

Soil Digest Modified USEPA 3050B Method

Weight 0.5g of dried, ground and well mixed sample into a microwave vessel. Record the exact weight.

Add 2.0mls Trace element grade Nitric acid and 2.0ml of 30% hydrogen peroxide.

Seal the vessels and vortex to ensure the acids and sample is well mixed. Load the vessels into the turntable and place into the microwave cavity.

Soil Digest Program

Ramp to 90°C over 15mins, hold for 5 mins

Ramp to 200°C over 15 minutes, hold for 20 mins

The internal temperature of the vessels is continually monitored by 2 Infrared sensors in the bottom of the microwave cavity.

The cooled samples are uncapped and made up to 25ml using MilliQ water. Soil samples are filtered using Whatman 52 filter paper.

Microwave Digestor

The **CEM MARS Xpress** has an operator selectable output of 0 – 1600 watts \pm 15% (by IEC (International Electrical Conference) method). Microwave energy is used to heat samples in a closed vessel microwave system. A sample is placed inside a Teflon PFA® and kevlar shielded vessel usually with conc. nitric acid. Once in the MARS Xpress, the samples are subjected to rapid heating and elevated pressures, causing the sample to digest or dissolve in a short time. Up to 40 samples can be digested at a time. A blank and reference sample are included in each batch.

Appendix C

Macroinvertebrate Total Abundance Data:

Macroinvertebrate total abundance data from each sample site and research trip. An MCI score of 10 represents highly sensitive taxa, and an MCI score of 1 represents taxa that are highly tolerant to pollution.

Grandview Creek site one

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	20	2	3
<i>Aphrophila</i> (5)	0	2	10
<i>Archichauliodes</i> (7)	13	0	7
<i>Austrosimulium</i> (3)	18	4	11
<i>Beraeoptera</i> (8)	0	8	2
<i>Confluens</i> (5)	0	10	0
<i>Deleatidium</i> (8)	20	29	14
<i>Eiseniella</i> (1)	0	1	3
<i>Hexatomini</i>	0	0	1
<i>Hudsonema</i> (6)	0	0	1
<i>Hydranidae</i> (8)	2	3	0
<i>Hydrobiosis</i> (5)	0	2	2
<i>Lobodiamesa</i> (5)	0	62	0
<i>Neppia</i> (3)	0	0	1
<i>Olinga</i> (9)	31	12	40
<i>Orthocladiinae</i> (2)	0	20	8
<i>Potamopyrgus</i> (4)	2	17	34
<i>Pycnocentroides</i> (4)	0	0	2
<i>Spaniocercoides</i> (-)	1	0	0
<i>Stenoperla</i> (10)	2	0	6
<i>Zelandobius</i> (5)	1	0	0

Grandview Creek site two

Genus (MCI Score)	Abundance – Winter	Abundance – Spring	Abundance – Summer
<i>Aphrophila</i> (5)	0	0	2
<i>Archichauliodes</i> (7)	2	1	1
<i>Austrosimulium</i> (3)	18	94	12
<i>Coloburiscus</i> (9)	0	0	1
<i>Confluens</i> (5)	0	44	12
<i>Deleatidium</i> (8)	67	119	4
<i>Eiseniella</i> (1)	6	1	4
<i>Elmidae</i>	0	0	1
<i>Hudsonema</i> (6)	1	1	6
<i>Hydraenidae</i> (8)	0	1	0

<i>Hydrobiosis</i> (5)	9	14	1
<i>Lobodiamesa</i> (5)	0	24	0
<i>Oligochaetes</i> (1)	1	0	0
<i>Olinga</i> (9)	9	6	3
<i>Orthocladinae</i> (2)	0	8	4
<i>Potamopyrgus</i> (4)	0	0	6

Lagoon Creek site one

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	27	23	18
<i>Aphrophila</i> (5)	0	1	2
<i>Archichauliodes</i> (7)	0	2	4
<i>Austrosimulium</i> (3)	0	2	7
<i>Coloburiscus</i> (9)	11	8	11
<i>Confluens</i> (5)	9	0	3
<i>Deleatidium</i> (8)	68	13	41
<i>Eiseniella</i> (1)	2	1	3
<i>Elmidae</i> (6)	0	0	7
<i>Hudsonema</i> (6)	1	1	2
<i>Hydrobiosis</i> (5)	3	2	1
<i>Lobodiamesa</i>	0	22	0
<i>Neppia</i> (3)	2	1	0
<i>Olinga</i> (9)	25	15	3
<i>Orthocladinae</i> (2)	0	0	3
<i>Potamopyrgus</i> (4)	0	3	6
<i>Pycnocentrodes</i> (5)	0	2	1
<i>Spaniocercoides</i> (-)	1	1	0
<i>Stenoperla</i> (10)	7	2	5
<i>Zelandobius</i> (5)	5	3	1
<i>Zelandoperla</i> (10)	3	3	0

Lagoon Creek site two

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	5	1	0
<i>Aphrophila</i> (5)	0	1	0
<i>Austrosimulium</i>	0	1	1
<i>Deleatidium</i> (8)	13	44	57
<i>Eiseniella</i> (1)	6	2	10
<i>Elmidae</i> (6)	0	4	4
<i>Hexatomini</i> (5)	0	0	1
<i>Hudsonema</i> (6)	0	1	0
<i>Hydraenidae</i> (8)	0	0	1

<i>Hydrobiosis</i> (5)	0	1	3
<i>Lobodiamesa</i> (5)	0	1	0
<i>Oeconesidae</i> (9)	0	2	0
<i>Orthocladiinae</i> (2)	0	2	1
<i>Philorheithrus</i> (8)	1	0	0
<i>Potamoprygus</i> (4)	0	0	1
<i>Zelandobius</i> (5)	1	29	1
<i>Zelandoperla</i> (10)	0	0	2
<i>Zelandotipula</i> (6)	0	1	0

Lagoon Creek site three

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	2	3	0
<i>Aphrophila</i> (5)	0	1	0
<i>Austrosimulium</i> (3)	3	4	8
<i>Confluens</i> (5)	5	11	0
<i>Deleatidium</i> (8)	0	8	11
<i>Eiseniella</i> (1)	6	2	0
<i>Elmidae</i> (6)	3	17	3
<i>Eriopterini</i> (9)	0	1	0
<i>Hudsonema</i> (6)	1	1	1
<i>Hydraenidae</i> (8)	0	1	0
<i>Hydrobiosis</i> (5)	0	0	2
<i>Lobodiamesa</i> (5)	0	2	0
<i>Neppia</i> (3)	0	2	3
<i>Oeconesidae</i> (9)	0	2	1
<i>Olinga</i> (9)	2	7	0
<i>Orthocladiinae</i> (2)	0	0	7
<i>Philorheithrus</i> (8)	4	0	0
<i>Potamoprygus</i> (4)	25	85	12
<i>Pycnocentroides</i> (5)	0	0	7
<i>Scirtidae</i> (8)	0	10	2
<i>Stenoperla</i> (10)	1	4	1
<i>Zelandobius</i> (5)	8	13	0
<i>Zelandoperla</i> (10)	0	1	0
<i>Zelandotipula</i> (6)	1	1	0

Lagoon Creek site four

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	1	0	2
<i>Aphrophila</i> (5)	0	0	1
<i>Austrosimulium</i> (3)	1	14	3
<i>Confluens</i> (5)	0	34	0

<i>Deleatidium</i> (8)	64	226	60
<i>Eiseniella</i> (1)	1	7	4
<i>Elmidae</i> (6)	1	4	6
<i>Hudsonema</i> (6)	35	0	19
<i>Hydrobiosis</i> (5)	10	9	9
<i>Lobodiamesa</i> (5)	0	7	0
<i>Nuerochorema</i> (6)	0	5	1
<i>Olinga</i> (9)	23	10	1
<i>Orthocladiinae</i> (2)	0	0	1
<i>Polypedilum</i> (3)	0	0	1
<i>Potamoprygus</i> (4)	1	0	0
<i>Pycnocentria</i> (7)	0	22	15
<i>Pycnocentrodes</i> (5)	0	0	20
<i>Zelandobius</i> (5)	15	4	1
<i>Zelandoperla</i> (10)	1	2	0

Cameron Creek site one

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Austroperla</i> (9)	1	0	1
<i>Deleatidium</i> (8)	28	36	25
<i>Eriopterini</i> (9)	1	0	0
<i>Eiseniella</i> (1)	9	1	2
<i>Elmidae</i> (6)	1	0	0
<i>Gyraulus</i> (3)	0	0	1
<i>Hudsonema</i> (6)	2	0	1
<i>Hydraenidae</i> (8)	0	0	3
<i>Hydrobiosis</i> (5)	1	0	1
<i>Nannochorista</i> (7)	0	0	1
<i>Neurochorema</i> (6)	0	2	0
<i>Neppia</i> (3)	3	0	1
<i>Orthocladiinae</i> (2)	0	0	5
<i>Ostracoda</i> (3)	0	0	2
<i>Philorheithrus</i> (8)	1	6	4
<i>Potamoprygus</i> (4)	0	0	8
<i>Oeconesidae</i> (9)	1	0	0
<i>Spaniocerca</i> (8)	0	0	1
<i>Zelandobius</i> (5)	3	1	1
<i>Zelandotipula</i> (6)	0	1	0

Cameron Creek site two

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	1	1	1
<i>Aphrophila</i> (5)	0	0	3

<i>Austrosimulium</i> (3)	0	0	4
<i>Deleatidium</i> (8)	0	26	79
<i>Eiseniella</i> (1)	2	0	4
<i>Elmidae</i> (6)	0	1	5
<i>Hudsonema</i> (6)	0	0	1
<i>Hydrobiosis</i> (5)	2	1	7
<i>Nannochorista</i> (7)	1	0	0
<i>Neppia</i> (3)	3	3	9
<i>Nothodixa</i> (4)	0	0	1
<i>Oeconesidae</i> (9)	0	0	1
<i>Orthocladiinae</i> (2)	2	0	0
<i>Philorheithrus</i> (8)	0	1	4
<i>Potamoprygus</i> (4)	34	156	104
<i>Scirtidae</i> (8)	0	1	0
<i>Spaniocerca</i> (8)	0	0	1
<i>Tiphobiosis</i> (6)	0	0	1
<i>Zelandobius</i> (5)	6	1	0

Cameron Creek site three

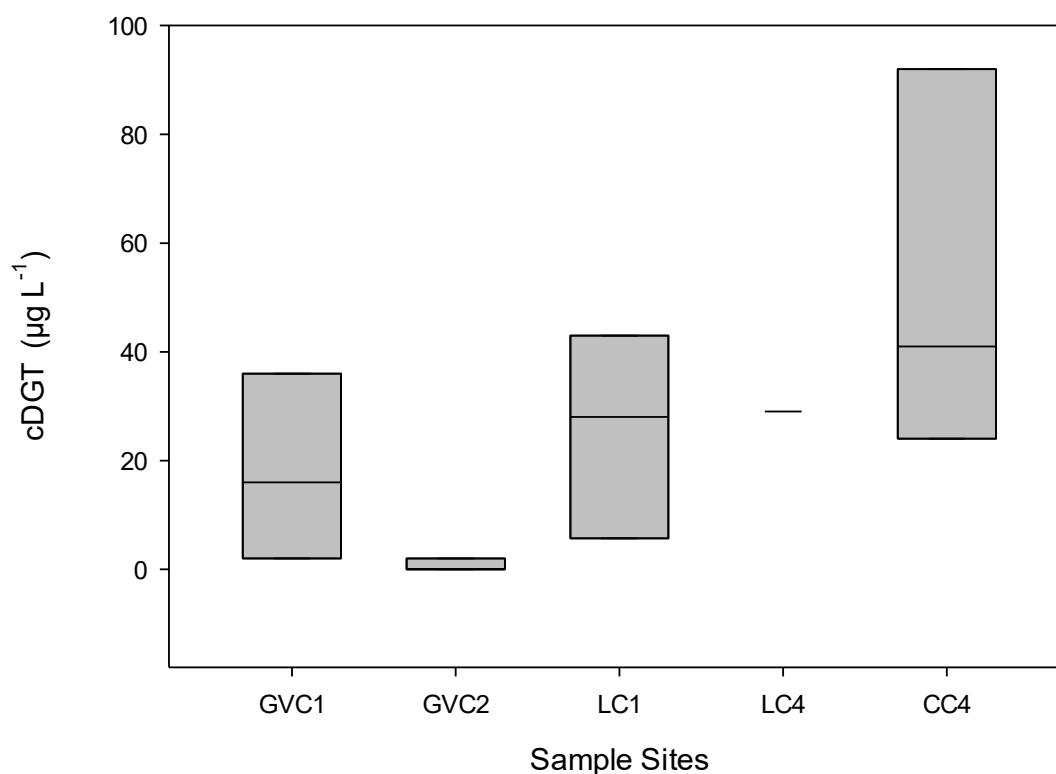
Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Aoteapsyche</i> (4)	0	1	2
<i>Aphrophila</i> (5)	0	0	1
<i>Austroperla</i> (9)	0	0	1
<i>Austrosimulium</i> (3)	0	12	4
<i>Deleatidium</i> (8)	4	10	38
<i>Eiseniella</i> (1)	0	2	5
<i>Elmidae</i> (6)	0	0	1
<i>Halticoperla</i> (-)	0	0	13
<i>Hudsonema</i> (6)	0	1	0
<i>Hydrobiosis</i> (5)	0	1	13
<i>Lobodiamesa</i> (5)	0	18	0
<i>Neppia</i> (3)	0	0	2
<i>Nothodixa</i> (4)	4	2	4
<i>Potamoprygus</i> (4)	18	105	91
<i>Pycnocentrodes</i> (5)	0	0	1
<i>Orthocladiinae</i> (2)	0	0	2
<i>Spaniocerca</i> (8)	0	0	7
<i>Zelandobius</i> (5)	16	47	3

Cameron Creek site four

Genus (MCI Score)	Abundance – July	Abundance – September	Abundance – December
<i>Deleatidium</i> (8)	0	3	1
<i>Eiseniella</i> (1)	3	1	2

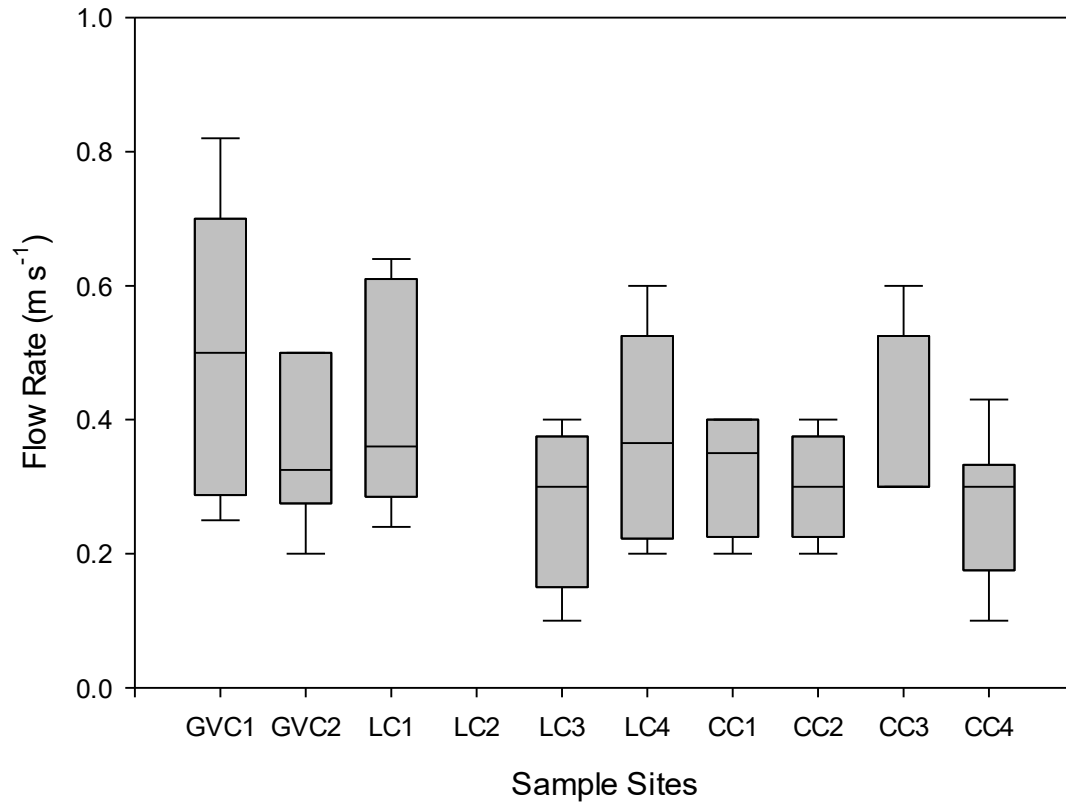
<i>Gyraulus</i> (3)	0	0	5
<i>Hydrobiosis</i> (5)	0	1	0
<i>Oligochaetes</i> (1)	0	30	0
<i>Nothodixa</i> (4)	0	0	1
<i>Orthocladinae</i> (2)	2	0	0
<i>Potamopyrgus</i> (4)	5	129	17
<i>Scirtidae</i> (8)	0	0	1

Appendix D



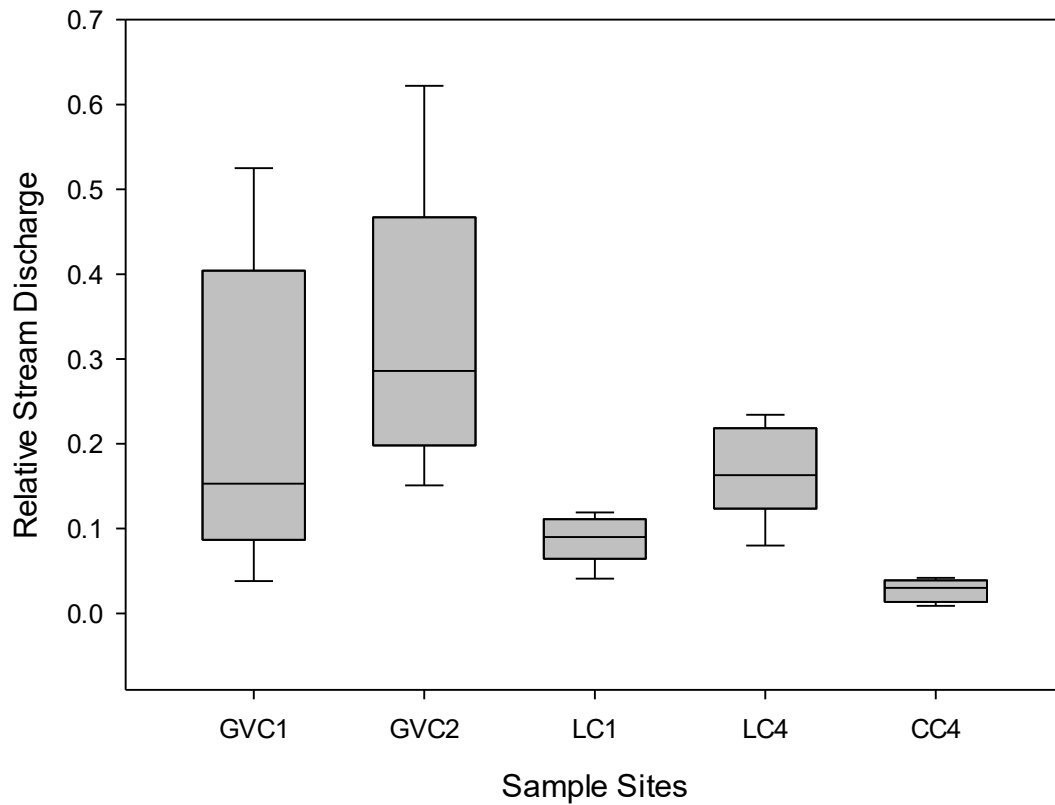
In-stream cDGT phosphorus concentrations at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. Each bar represents the cDGT concentrations for a single sampling event. Two samples at site GVC2 were below the method detection limit ($1.5 \mu\text{g L}^{-1}$), and two samples at site LC4 failed to provide any data. Samples were collected over the duration of three research field trips and the data combined.

Appendix E



The flow rate of the stream water at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The flow rate at LC2 was too low to measure. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. Samples were collected over the duration of three research field trips and the data combined.

Appendix F



The relative stream discharge at sample sites, with each site presented from higher to lower elevation sampling locations left to right in the Grandview Creek, Lagoon Creek and Cameron Creek catchments. The boxes show the first and third quartiles with median in the centre. The whiskers show the maximum and minimum values. Samples were collected over the duration of three research field trips and the data combined.

Appendix G

Rainfall Data:

Mt. Grand Station rainfall data, recorded during periods of field research. Rainfall data was taken from the weather station above Hospital Gully at 620m. The tables below show rainfall for the seven days prior to, and during each research trip. Sampling days have been highlighted in red.

Research Trip One

date	Rain (mm)
27-Jun-16	0
28-Jun-16	0
29-Jun-16	0
30-Jun-16	0
1-Jul-16	0
2-Jul-16	0
3-Jul-16	0
4-Jul-16	0
5-Jul-16	0
6-Jul-16	0
7-Jul-16	0

Research Trip Two

date	Rain (mm)
13-Sep-16	0
14-Sep-16	0
15-Sep-16	0
16-Sep-16	2.4
17-Sep-16	0
18-Sep-16	0
19-Sep-16	0
20-Sep-16	0
21-Sep-16	0
22-Sep-16	0
23-Sep-16	0

Research Trip Three

date	Accumulated Rain (mm)
7-Dec-16	13.4
8-Dec-16	4.2
9-Dec-16	0
10-Dec-16	0
11-Dec-16	0
12-Dec-16	1.6
13-Dec-16	0
14-Dec-16	1.4
15-Dec-16	0
16-Dec-16	0
17-Dec-16	0

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